

Identification and Evaluation of Performance Indicators for Abalone Fisheries

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1 Non-Technical Summary

2012/225: Identification and Evaluation of Performance Indicators for Abalone Fisheries

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	OBJECTIVES
1	Determine, document and review the Performance Indicators (PIs), related stock assessments and fishery management objectives used in the abalone fisheries of Australia and similar fisheries worldwide.
2	Identify in close collaboration with abalone Industry, Management, and researchers, a suite of fishery assessment PIs that facilitate assessments against the management objectives for abalone fisheries.
3	Where possible, evaluate the fishery assessment PIs against known fishery performance.
4	Develop a National Management Strategy Evaluation framework that can be adapted to represent different abalone fisheries from the various jurisdictions in southern Australia.
5	Identify, using the PIs determined in Objective 1, a suite of Management Strategies (i.e. unique combinations of data, PIs and decision rules) that aim to achieve the fishery objectives identified in objective 1).
6	Use the Management Strategy Evaluation framework (from objective 4), to assess the relative effectiveness of the alternate Management Strategies (from Objective 5) to achieve the fishery objectives, in the face of multiple sources of uncertainty and spatial variation in data availability and quality.

Outcomes achieved to date

1. The South Australian management plan has received substantial benefit from the international review of management objectives and PIs.
2. Implementation issues with the new harvest strategy in South Australia have been clarified by the use of the MSE framework and modifications have been proposed.
3. The output of the MSE framework will be assisting with the design of the multi-criterion decision analysis management framework to be introduced into the Tasmanian abalone fishery.
4. Current (2013/2014) discussions regarding the most appropriate combinations of LML and TACs are being illuminated through use of the MSE framework. For example, there are current moves to increase the LML on the south-west coast from 140mm to 145mm
5. In Western Victoria, the MSE framework has had an initial application to the virus affected stocks and rebuilding plans that include some fishing. It assisted in selecting an appropriate annual harvest rate combined with a given LML, that permitted some fishing while retaining a sufficiently precautionary rebuilding rate.

1.1 Stage 1: Performance Measures

Australian abalone (Family: Haliotidae; Genus: Haliotis) fisheries, based predominantly on greenlip (*Haliotis laevis*) and blacklip (*H. rubra*) abalone, appear stable and sustainable relative to abalone fisheries elsewhere in the world. In part, this is due to the management arrangements in place in each State that include limited entry, legal minimum lengths, spatial management and individually transferable quotas (ITQs) controls, underpinned by annual fishery assessments based on a broad range of fishery-dependent and fishery-independent data and guided by formal management plans.

A key feature of recent management plans has been the development of fishery performance indicators (PI = performance measure or PM), and, in some cases, associated target and limit reference points. For such PM to be useful for management they need to be robust to natural biological variability and related uncertainties. Thus, empirical PMs must operate within associated harvest control or decision rules to manage fisheries towards whatever objectives are chosen for them.

This project took a two-stage approach to addressing these requirements. First, performance indicators for abalone fisheries, in the context of the management objectives were reviewed and then examined qualitatively by expert panels and against fishery data. The second stage was a formal, quantitative analysis of current management strategies undertaken using a procedure called Management Strategy Evaluation (MSE). A Management Strategy is the combination of data, the performance measure (= assessment used), and decision rule used following the assessment, and is becoming broadly recognized as the accepted method for conducting such testing. This is because it is necessary to test the performance of complete management strategies, rather than PMs in isolation from either the quality and quantity of their associated data, or the decision rules with which they would be used. MSE provides the additional benefit of testing PI in a simulation environment, rather than rely on empirical testing of PIs in situ.

Management objectives among the Australian state-based abalone fisheries encompassed biological (includes ecological and environmental), economic, governance (management), and social categories, with the biological objectives dominating numerically and providing the principal management direction. A diverse range of PIs are used for the assessment of Australian state-based abalone fisheries, with most relating to assessing fishery performance against biological objectives and, almost exclusively, those relating to sustainability rather than ecosystem integrity. The PIs are obtained from a broad range of sources including fishery-dependent and fishery-independent data and outputs from numerical models, with catch rates being the most common PI used. Several potential, novel PIs are under development in Australia, most notably those based on spatial indices of stock status and industry knowledge and perception. The latter would provide a formal mechanism for incorporating ‘diver assessments of stock status’ into harvest strategies, harvest control rules and TACCs and would overcome the problems with the existing ad hoc and informal inclusion of this information.

The expert-panel workshops undertaken in this study provided the first step to assessing PI suitability. Overall, all expert panels identified three PIs – raw CPUE ($\text{kg}\cdot\text{hr}^{-1}$), proportions of large and small length classes in the commercial catch and diver assessment of stock status – as very useful and for the biological PIs, 23 achieved a rating of $\geq 75\%$ of the maximum possible score therefore being considered “preferred” PIs. Although

the assessment process was rapid, expert-panel approaches are entirely qualitative and “opinion driven”, with the potential for PIs to receive a ranking (i.e. high or low) that is inconsistent with their historical or potential performance.

When tested against available data, numerous PIs changed through time in a manner consistent with declines in fishery performance and reflective of reductions in legal-size abalone abundance. This was particularly evident for PIs at Cowell and for mean size, median size, proportion small and proportion large in Waterloo Bay. Although the key weakness of this approach was that few of the data were from a fishery operating commercially with a consistent number of experienced fishers targeting a familiar species over an extended period of time, (4 years), the initial, quantitative analyses highlighted the strength of those based on commercial-catch-sampling data, along with the need to explicitly consider (1) the suite of PIs that most closely match management objectives; (2) sensitivity of PIs to detect change, particularly their ability to measure decreases in abundance prior to stock collapse; (3) minimum data requirements; (4) factors that bias data; and (5) statistical methods employed.

1.2 Stage 2: MSE Testing of Harvest Control Rules

A spatially explicit, size-structured, abalone population simulation model has been developed that is capable of simulating a single abalone population, a small collection of abalone populations representing a statistical assessment unit (SAU), or even numerous populations representing an abalone fishing zone (which contains multiple SAUs, which contain multiple populations). The MSE framework is capable of simulating and following numbers at size and in the catch from each of the constituent populations in its definition. In addition, it follows biomass at size, catch rates, catches, the number of recruits, and most other statistics relating to the population dynamics of the various populations. The simulated populations can be fished at any required Legal Minimum Length, and TAC, and both can be modified through time as necessary. The software is not particularly user friendly but it is structured so that adding new harvest control rules and changes to the dynamics is now relatively simple and as further development proceeds, attempts will be made to simplify the interface to its required inputs and to automate the analysis of its outputs so that more people can actively use it for themselves to explore management options.

In order to condition the model so that it resembled a known fishing zone it was necessary to conduct a large number of analyses of basic biological properties using a Tasmanian abalone biology database built up over the last few decades. This is a significant step, requiring significant allocation of time in order to allow the model to be run efficiently, and simulate an abalone zone that was similar to Tasmania’s east coast. This also prepared the simulation framework for conditioning on any other abalone zone given the correct biological and fisheries data.

The current fishery assessments in Tasmania use a qualitative consideration of commercial catches and catch rates, with occasional reference to the length frequency of the catches from different areas. These performance measures were explored for Tasmania to determine how variable they are and whether they were useful as measures of the stock status. The high contrast that has been experienced in the Tasmanian fishery since the introduction of catch quotas in 1985, has led to periods of relatively light exploitation followed by high exploitation and then low exploitation again. Catches and effort

were not found to be informative of the stock but rather followed the management which derived from the informal consideration of catch rates and whatever other data was available each year. Diver perceptions are also important in the assessment process, forming part of the weight of evidence approach used in assessing fishery performance.

To test whether the use of catch rates was reasonable an effort was made to determine whether they were reflective of the stock dynamics. If catch rates reflect relative stock size then, the expectation is that catch rates would be influenced by annual harvests if the catch taken was large enough to alter the availability of exploitable biomass. If they were then if catches rose, the expectation would be that catch rates would fall after a time-lag that reflected the recruitment dynamics and growth dynamics of the population. Conversely, if harvest was reduced, then catch rates would be expected to eventually rise as more recruits entered and remained in the fishery without being taken. This inverse relationship was found for both the east and west coasts of Tasmania except that the optimum time-lag between CPUE and catches was seven years on the west coast and five years on the east. This finding is in contrast with the general assumption that abalone catch rates are not informative about the state of abalone stocks.

As well as exploring empirical performance measures, model based performance measures, such as fishing mortality rate, and level of spawning biomass, were also considered in the context of relatively simple surplus production models and more complex size-based integrated stock assessment models. Given the high contrast of the catch rates in Tasmanian fisheries it is not surprising that these models appeared very effective and generated very similar outcomes. The advantage of, and reason for retaining, the more complex size-based assessment model is that it permits explorations of the implications of changing the Legal Minimum Length (LML) as well as changing the Total Allowable Catch (TAC).

A key issue in the management of abalone stocks is setting the Legal Minimum Length. This invariably leads to controversy and increased concerns especially on the part of industry. The interactions and the assumed trade-offs between the TAC and LML were explored using the MSE simulation framework. The specific questions examined were how varying the LML affected the proportion of the mature biomass protected, and the consequences for the sustainable TAC that could be taken. Additionally the MSE explored the advantages and disadvantages of the Tasmanian Abalone Management Plan policy of ensuring the LML provides two years of protection (two year rule).

The maximum yield per recruit (an equilibrium concept) for blacklip abalone appears to occur at sizes a few millimetres smaller than the Biological Minimum Length (BML; zone average size at maturity plus two years' growth). For productive populations if the LML is close to the BML there is little difference in the potential yield and the proportion of the mature biomass protected brought about by small adjustments in the precise LML implemented. However, for less productive populations (with smaller BMLs), changes in potential yield and proportion of mature biomass protected were more sensitive to increases in LML. In other words it is more difficult to find a compromise LML that is effective across large geographical areas that contain regions of relatively low productivity, and pockets of highly productive fishing grounds. As most yield comes from the areas of relatively high production, adequately protecting these highly productive sites is important.

The two year rule appears to be a reasonable compromise when adopting a LML for a large area. When the LML approximates the average BML it prevents excessive under- and over-protection within a zone; fishing with a LML 5 – 10 mm smaller than the BML leads to a much greater risk of depletion for a given TAC, than fishing with an LML closer to the BML. Conversely, if the LML is set 5 – 10 mm above the BML then, especially for populations with smaller BML, there can be over-protection such that significant losses in potential yield will occur.

It can be recommended that any LML chosen should take into account the average size at maturity leaving a significant buffer before permitting fishing mortality. Alternatively a proxy, perhaps related to some fraction of the maximum observable size, could potentially be used instead.

The time taken for depletion to occur in simulated abalone stocks can be quite extended. In the simulations, dropping whole zones from about $0.4B_0$ to about $0.2B_0$ could take 40 years. However, in that time, some of the component populations within the zone had become so depleted that commercial fishing was no longer viable. If the fishing were occurring at levels well above the sustainable yields then depletion times could be much shorter.

Finally, the MSE simulation framework was used to compare and test the efficacy of two new harvest control rules (HCR) management processes that used CPUE. The first HCR used the gradient of recent changes in CPUE as a performance measure to determine whether the TAC should increase or decrease and the second set a target CPUE and the TAC changed depending on whether it was below or above the target, with the amount of change being affected by how far the current CPUE (the performance measure) was from the target. Both these HCR could be modified by the inclusion of a lower limit on the TAC, and both were implemented within a decision framework similar to a multi-criterion decision analysis framework. Questions asked of the HCR were whether they were capable of recovering a depleted stock, how they managed a stock being fished close to its Maximum Sustainable Yield (MSY), and whether they could manage an under-fished stock in a reasonable manner.

The capabilities of the HCR were tested by analysing the outputs from running 54 scenarios for each HCR, with each scenario being run with 100 replicates. The scenarios were defined using three initial TACs (above, approximately at, and below the zone's MSY), three initial depletion levels for the zone (above, about at, and below the level required to produce the MSY), and with the three LML. The simulated zone had a BML of 138 mm and had a MSY of about 630 tonnes at all LML. As expected from the LML vs TAC work, this was distributed as 628t at 127mm, 633 t at 132mm, and 632t at 138mm, with the most spawning biomass protected at the 138mm LML but the lowest CPUE at the MSY.

These 3 x 3 x 3 variables produced 27 combinations, which were doubled to 54 by including or not the option of a lower limit on the TAC.

The HCR that used the CPUE gradient performance measure, at best, generated management advice that led to a status quo but only for scenarios at or above MSY levels. Time lags between altering catches (TAC) and the effect of that on future CPUE were so long that they negatively affected the performance of this harvest strategy. It was es-

entially ineffective at providing timely management advice. Worse, it was unable to recover a depleted stock or optimize catches in a fishery starting from an un-depleted state. The time lags appear too long relative to the dynamics of the stock and lead to management responses occurring too many years after any decline to have sufficient positive effects. The over- and under-compensation observed with respect to catch levels simply took too long to feed through the dynamics of the stock to influence the CPUE that was the foundation of the HCR. These failures in the compensation led spawning biomass and catch rates to oscillate, in some cases with wide ranges between upper and lower levels at a frequency similar to those currently seen on the east coast. The oscillations in the catches were even greater when there was no lower limit on the TAC. The introduction of a lower limit on the TAC generally helped stabilize catches but also led to more rapid declines in spawning biomass and catch rates. Perturbing the TAC by a one off reduction in TAC at the introduction of the HCR did not influence the outcome in any positive way.

In its current form the CPUE gradient HCR cannot be recommended for use in the provision of management advice. Minor improvements were achieved through inclusion of asymmetry in the HCR response to the CPUE gradient performance measure such that increases in TAC were constrained to smaller steps and TAC decreases could occur in larger steps. When an asymmetric HCR was implemented it did not alter the behaviour with non-depleted stocks but it did add the capacity for the HCR to recovery depleted stocks. The inclusion of asymmetrical responses in HCR should be explored further.

The target CPUE HCR was far more effective at producing management advice that could permit the recovery of a depleted stock, and control the way a relatively unfished fishery might develop. Not surprisingly, the most stable outcomes arose from scenarios where the simulated zone was being fished close to the MSY at the state of depletion required to produce the MSY. However, both initial TACs above and below the MSY led to long period oscillatory outcomes. This appears to be a combination of the time-lags inherent in the dynamics of using CPUE to manage biomass but also those that follow from the switching behaviour that occurs when the current catch rate moves from below to above the target (or vice versa). It appears that this combination of factors leads to much longer period oscillations that with the CPUE gradient HCR.

While the target CPUE HCR could manage a depleted stock back to a healthier position it could take a long time and also lead to relatively small catches if there is no TAC limit imposed. Imposing a TAC limit certainly acts to maintain catches but it also acts to delay any recovery and this is exacerbated by increasing the LML. In many instances having a larger LML appears to make responses to change more variable and more extreme. If a TAC limit is used and set too high (equivalent to holding up catches in a depleted stock) a stock can be held down so that no rebuilding occurs and it continues along at very low catch rates.

The multi-criterion decision analysis used in South Australia and proposed for Tasmania may have some issues with the implementation of the scoring system which can affect the responsiveness of the HCR to changes in the performance measure they are based upon. For example, when the suggested scoring system is implemented within the target CPUE HCR the management advice it generates tends to lead the fishery to stabilize above the target. However, solutions have been suggested and this can be corrected or at least fully understood for each HCR.

This project has successfully developed an MSE simulation framework for testing specific performance measure and harvest control rule strategies for use in abalone fisheries. Development of the MSE is complete, and can now be adapted for different jurisdictions, or to test specific scenarios of interest to management.

KEYWORDS

Abalone; Abalone Management Objectives; Management Strategy Evaluation; Simulation of Abalone Management; Performance Measures; Performance Indicators;

2 Acknowledgments

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3 Background

Challenge 1:

- Improve the sustainability of natural resources supporting wild-catch and aquaculture.
- improve governance, policies, and methods in wild-catch management;
- in particular review and assess abalone fishery stock assessment performance indicators, assessments and management objectives, where after conduct management strategy evaluation to select among a range of the most likely fishery performance measures used to assess abalone fisheries.

Abalone (Family: Haliotidae; Genus: Haliotis) are highly prized and valuable marine gastropods inhabiting near-shore reefs (Day and Shepherd 1995) from the shallow subtidal zone to depths around 400 m (Geiger 1998, 1999). Historic artisanal fisheries (Ino 1966, Tegner 1988, Guzmán del Prío 1992, Buchanan 1988) were followed by large-scale commercial abalone fisheries across nine geographic regions. These persisted until the late 1980's, during which global production peaked at about 15,000 t. yr⁻¹ (Gordon and Cook 2001). Thereafter, production from most fisheries declined rapidly, with many of the major fisheries in the world having collapsed (Shepherd and Baker 1998, Kojima 1981, Davis et al. 1996, 1998, Parker et al. 1992, Haaker et al. 1996, Tegner 2000, Tegner et al. 2000a, Farlinger and Campbell 1992, Sloan and Breen 1988, Tarr 2000).

In general contrast to the broad global trends, Australian abalone fisheries have fared comparatively well. These fisheries, based predominantly on greenlip (*Haliotis laevis*) and blacklip (*H. rubra*) abalone, appear comparatively stable and sustainable. Management arrangements vary among States, but typically include a range of input (e.g. limited entry) and output (e.g. Legal Minimum Lengths (LML) and spatially-managed (Zonal) individually transferable quotas (ITQs)) controls, underpinned by fishery assessments based on a broad range of fishery-dependent and fishery-independent data and guided by formal management plans (e.g. Zacharin 1997, Nobes et al. 2004, Tasmanian Abalone Fishery Revised Policy Paper, NSW Abalone Fishery Management Plan, Victorian Abalone Fishery Management Plan, WA Fisheries Management Paper No. 204). Management plans aim to ensure future sustainability of these fisheries through contemporary management approaches.

A key feature of management plans in recent years, partly in response to changes in biodiversity conservation legislation requiring all Australian export fisheries to demonstrate ecological sustainability, has been the development of fishery performance indicators (PI), and, in some cases, associated target and limit reference points. Typically, such performance measures are aggregated into four classes – biological, social, economic and environmental – that collectively inform the decisions upon which management of the fishery is largely dependent.

While a range of PI for finfish fisheries are well accepted as tools for fishery management (Caddy 1998), similar levels of agreement have not yet been reached (or attempted) for abalone fisheries. Consequently, a multitude of PI are used in the management of abalone fisheries in Australia (Gorfine et al. 2001). Notably, this diversity has, in part, resulted in a common recommendation from CDEH for State fishery agencies to collectively pursue a national approach to developing, adopting and reviewing these indicators.

To ensure appropriate management and, consequently, future sustainability of these fisheries, these PI must be robust. Thus, for biological PI, they must provide clear and timely indications of variation in abalone abundance and/or population structure. Hence, they must be sensitive to and effective at detecting change. Without this capability, they will fail to identify Zones/Regions/Reefs where the resource may be heading towards being overfished, or, alternatively, where the resource could sustain additional fishing pressure. Currently, the utility of the multitude of PI used in the management of Australian abalone fisheries to act either as an ‘early warning signal’ or as an indicator of improving resource status for these fisheries is poorly understood.

Archiving over the lack of consistency in approach among States and the poor understanding of the applicability of the PI used, is the topical issue of aligning the scale of fishery assessment and management with the scale of biological stocks (Hilborn 1995). In line with this, two current FRDC-funded projects (2004/019 – Towards optimising the spatial scale of abalone fishery management, and 2005/024 – Abalone Industry Development: local assessment and management by industry) are challenging abalone management-‘dogma’ and current assessment approaches. Future abalone fishery assessment and management is likely to occur at finer spatial scales (e.g. sub-zones, fishing areas, map-codes or reefs) than that currently broadly employed (i.e. zone, region). This change also necessitates reconsideration of fishery performance measures.

Thus, the need to develop spatially-relevant, defensible assessments for abalone that have a predictive capacity is very great. In this project, we propose a two-stage approach to addressing these needs. The first stage focuses on reviewing stock assessment performance indicators and stock assessments for abalone fisheries, in the context of the management objectives, and to examine these qualitatively through their evaluation by expert panels and against fishery data. Subsequently, the second stage focuses on assessing the most promising performance indicators more quantitatively and more formally through undertaking a procedure called Management Strategy Evaluation. This is becoming broadly recognized as the accepted method for conducting such testing. A Management Strategy is any combination of the data collected, the performance measures used (= assessment used), and the decision rules used following the assessment. It is the combination of all three of these components that determines the effectiveness of a management strategy. Even if a performance measure existed which had the potential to represent the exact status of an abalone stock, without the appropriate data and without appropriate management actions following the assessment, the management of the stock would still not be optimal. It is therefore necessary to test the performance of complete management strategies. In other words, it is not possible to test the relative performance of performance measures in isolation from the quality and quantity of their associated data or away from the context of the decision rules with which they would be used.

Some performance measures (PM) can be used to characterize a stock’s status by comparing the estimates of the measure against different reference points – a Limit Reference Point (LRP) and a Target Reference Point (TRP). The LRP should represent a state of the stock to be avoided and if the PM breaches the LRP then stringent management action is usually required to move the stock back towards the TRP, which is defined as a desirable state for the stock. Ideally, given a PM with associated LRP and TRP there should also be a Harvest Control Rule or Decision Rule, which defines the management actions that should be undertaken depending upon whether the PM is close to the TRP or the LRP. An example of a performance measure might be the catch per unit of effort within a

given reference area with LRP and TRP calibrated for the given reference area. Management actions could include such things as changes in the Total Allowable Catch (TAC) from an area, changes to a season length, and changes to the legal minimum length. An advantage of such an approach is that the assessment process and management advice resulting from the assessment would be transparent and clear to all stakeholders. However, this approach would only be acceptable if general confidence was held in the assessment process, or in other words, there was confidence that the performance measures used provided a workable representation of the status of the stock being assessed and that the management procedure led to sensible outcomes for the fishery and the stock. Presently, the effectiveness of all of the performance measures used in Australian abalone fisheries, whether from formal assessment models or from the more informal approaches, is unknown; although none are considered to provide a clear characterization of a stock's status.

Thus, while both formal and informal assessment approaches exist, there remains a real need to determine how effective stock assessment methods (= performance measures) really are at providing management advice that works. For example, by providing false confidence, ineffective stock assessment models can be more dangerous for resource sustainability than not using a stock assessment model. If ineffective performance measures are trusted then the wrong management advice can be followed leading to stock declines and underperformance.

The introduction of such formal management arrangements as the agreed use of performance measures with associated LRPs and TRPs combined with pre-defined decision rules is an approach which is relatively new to Australia. For example, presently, the Australian Fisheries Management Authority (AFMA) is attempting to define performance measures, LRPs, and TRPs for all Commonwealth fisheries in order to bring both credibility and transparency to the management of Commonwealth stocks. Obviously for such management schemes to be workable the performance measures and reference points selected must have a number of properties:

- 1) They must be directly relevant to each fishery concerned (that is, there should be no default standard set but each fishery should have a custom set developed for its optimal management). For abalone this means that different jurisdictions may well use different management strategies to match their own situations and fisheries.

- 2) They must be estimable with sufficient accuracy so they can form the basis of clear management advice. For example, if stock assessment models can only estimate harvest rates with very low precision then LRPs and TRPs should not be based upon harvest rates or fishing mortality rates – because managers would always be uncertain whether the target had been attained or the limit had been exceeded.

- 3) They must reflect the biology of the species concerned. For example, there is no point in imposing a LRP demanding that the juvenile abundance be greater than some minimum where, as in blacklip abalone, the juveniles tend to be cryptic and sensible estimates are not possible.

Management Strategy Evaluation (MSE) is regarded as the best available approach for contrasting and comparing alternative management strategies. In each fishery, each different combination of data collection, stock assessment/estimation of performance

measures, and decision rules leading to management actions, is termed a Management Strategy. There can be numerous alternative management strategies developed for any single fishery. For example, 1) the data collected could be the daily catch rates across all divers in a zone for the whole season; 2) The assessment or estimate of the performance measure could be the geometric mean catch rate and variation about this figure is calculated, and 3) a decision rule could be that a 30% decline in the geometric mean across a two year period would lead to a 10% drop in the TAC. All three steps would constitute a management strategy. If the decision rule happened to be a 20% drop in TAC then the three combined would constitute a separate and different management strategy. A very significant question to be answered is which of these management strategies are most effective at achieving the management objectives selected for a fishery (especially in the context of uncertain data and delays in management responses)?

A simpler way of putting this would be to ask what data and performance measures combined with decision rules can be used to provide effective management advice? This question has been asked many times before, for example at a National Abalone Performance Indicator Workshop held at the South Australian Aquatic Sciences Centre on 25th July 2002, and at many meetings since. Despite all of this discussion both formal and otherwise, with abalone there have been no significant advances made on distinguishing effective performance measures from ineffective ones. For example, catch rates are known to be poorly representative of the status of abalone fisheries but such data are still routinely used in abalone stock assessments. Sometimes catch rates appear to be informative and sometimes not; the difficulty is in determining under what conditions they are useful and when are they dangerously mis-informative. Part of the reason for this lack of advancement is that making such comparisons is extremely difficult to do in practice, especially for abalone fisheries. Unfortunately, it appears the only way to be certain empirically about whether a particular performance measure is ineffective is if a fishery collapses. Just because a fishery appears to be progressing well does not mean that the performance measures being used are either effective or efficient. For example, the Tasmanian eastern zone abalone fishery seriously declined from 1999/2000 to 2004. Despite the initial serious decline it took until 2002 for the TAC to be reduced from 1190 t down to 857.5 t, this was followed in 2004 by a further reduction down to 770 t. This was a drop of about \$16.5 million in export value. The delay in management action in response to the decline in the eastern zone stock put the whole stock at risk and lowered catch rates to historical lows. This illustrates why there is a need to be able to assess the status of the stocks in a defensible manner and enable more timely management advice. It remains unknown what combination of performance measures and decision rules will optimize the trade-off between maximum harvest and stock sustainability. For example, if the management objective for Tasmania's eastern zone was to return the TAC to a total of 1,000 tonnes, we could use an MSE to determine whether it would be better to attempt that objective in a few large steps or with more but smaller steps.

An advantage of using MSE rather than experimentally manipulating real fisheries is the obvious one of being able to explore options that would have unacceptable risks with a real fishery. Truly experimenting with the management of a real fishery would not only be risky in practice it would also be extremely slow, especially if management proceeded cautiously. The best option is to use MSE to search for those management strategies which are robust to uncertainty in available data and in our knowledge of the stock dynamics. In fact there are many sources of uncertainty in resource management. When generating management advice one is aware of uncertain data, uncertain knowledge about

the dynamics of the stock involved, uncertainty about the future distribution of effort, uncertainty about the biology of the species concerned, and finally, there is uncertainty about how management decisions are implemented. MSE involves a simulation framework that considers the whole management system. It consists of a Virtual Fishery Simulation (or Operating Model) that is regarded as representing the accepted as “true” underlying dynamics of the resource and the fishery. It is best practice to use multiple or different Virtual Fishery Simulations so as to capture uncertainty about the true underlying dynamics of the stock. The Virtual Fishery Simulation also includes methods for generating the types of data usually collected from each fishery. The MSE framework includes the different assessment procedures and performance measures that are used to analyse the various fishery or monitoring data generated by the Virtual Fishery Simulation. The assessment procedures that estimate the performance measures are only “aware” of the generated data not the underlying dynamics of the assumed true Virtual Fishery Simulation. Finally, there are sets of decision rules that interpret the results from the performance measure estimates and generate management advice. This modelled management advice is fed back into the Virtual Fishery Simulation where it can obviously influence the dynamics of the virtual stocks being managed. In this way different management strategies can be simulated and the predicted outcomes from the assessments can be compared with the “true” situation from inside the Virtual Fishery Simulation. By including a wide range of uncertainties into the simulated data the MSE process can identify those management strategies and performance measures that are most robust to uncertainty and that enable management to best achieve its objectives. Developing and conducting a full MSE may take significant time (years) but would still provide for many more comparisons than could be contemplated with a real fishery.

In addition to the standard performance measures (some refer to these as performance indicators) there are many opportunities for developing new and potentially more sensitive performance measures, better suited to characterizing the stock status of species which are difficult to age. For example, in the Tasmanian abalone fishery, there are some areas where the fishing intensity is very high. A potential performance measure that will be investigated is to characterize the proportion of new recruits above the legal minimum size that survive for longer than one year. This should relate back to catch rates and if a minimum catch rate for an adequate economic return can be determined then Limit Reference Points based on survivorship should be informative to management. This and other potentially workable performance measures need to be investigated to determine their sensitivity to uncertainty in available data and erroneous representations of the dynamics of growth and recruitment. In order to conduct such investigations requires the development of a general Management Strategy Evaluation (MSE) framework. This would need to be general for it to be able to simulate the dynamics of abalone stocks at very different spatial scales and as representing conditions in different jurisdictions. By developing a MSE framework that would also be suitable for size-based models it will be possible to examine the functionality and sensitivities of different performance measures and identify those procedures robust to uncertainty in both data and assessment method. This may be especially important to abalone, which exhibit great variation from site to site. A question to be answered is: How well do any performance measures represent stock dynamics when a harvested species is as patchy and variable in its properties as abalone are around southern Australia? Without answering this question any set of performance measures (or stock assessment model) used with abalone will always be vulnerable to charges of not being representative of a particular area.

4 Need

The absence of reliable performance indicators (PIs) for Australian abalone fisheries has limited stock assessments to an informal, ad hoc review framework, (overly conservative management versus unrealistic optimism from Industry). Without a formal assessment framework based on effective PIs, with target (TRP) and limit reference points (LRPs) combined with clear decision rules, abalone assessment and management will continue to be exposed to decisions prejudiced by opinion rather than fact. Searching for effective PIs for assessing abalone fisheries has continued for years but they remain untested. A need remains to identify an array of informative PIs for abalone fisheries. This requires a re-consideration of PIs (both current and those being developed) and their formal testing in a Management Strategy Evaluation (MSE) framework. The Tasmanian Abalone Strategic Research Plan gives high priority to research into MSE and the development of TRPs and LRPs. NSW, Victoria, and Tasmania all listed MSE of PIs, LRPs and TRPs, as a high priority at the 2006 National R&D Workshop. South Australia places a high priority on the development of a new management plan for abalone, which requires the development of informative PIs. Most recently, the Draft National Abalone Health Work plan recommends conducting a MSE to compare likely outcomes following the viral outbreak in Western Victoria. Failure to find PIs and management strategies that will operate with different Australian abalone fisheries, constitutes a significant threat to their ongoing sustainability. If ad hoc assessments and their associated risks are to be avoided the debate over PIs needs to stop and the generally accepted method of MSE, needs to be developed for abalone fisheries. Without developing this predictive capacity, stock assessments will remain ad hoc and subject to considerations other than finding the optimum trade-off between maximizing the product value while minimizing the risk to sustainability.

5 Objectives

- 1) Determine, document and review the Performance Indicators (PIs), related stock assessments and fishery management objectives used in the abalone fisheries of Australia and similar fisheries worldwide.
- 2) Identify in close collaboration with abalone Industry, Management, and researchers, a suite of fishery assessment PIs that facilitate assessments against the management objectives for abalone fisheries.
- 3) Where possible, evaluate the fishery assessment PIs against known fishery performance.
- 4) Develop a National Management Strategy Evaluation framework that can be adapted to represent different abalone fisheries from the various jurisdictions in southern Australia.
- 5) Identify, using the PIs determined in Objective 1, a suite of Management Strategies (i.e. unique combinations of data, PIs and decision rules) that aim to achieve the fishery objectives identified in objective 1).
- 6) Use the Management Strategy Evaluation framework (from objective 4), to assess the relative effectiveness of the alternate Management Strategies (from Objective 5) to achieve the fishery objectives, in the face of multiple sources of uncertainty and spatial variation in data availability and quality.

6 Methods

This project took a two-stage approach. First, stock assessment performance indicators and stock assessments for abalone fisheries, in the context of the management objectives were reviewed and then examined qualitatively by expert panels and against fishery data. Subsequently, the second stage was a formal, quantitative analysis undertaken using a procedure called Management Strategy Evaluation. Stage 1 was led by Dr Stephen Mayfield with the work being undertaken by Dr Mayfield, Dr Rowan Chick, with the assistance of Dr Maria Jednesjo. This component of the project was undertaken at SARDI in Adelaide. Assoc. Prof. Malcolm Haddon led Stage 2 with the work undertaken by Dr Haddon and Dr Fay Helidoniotis of CSIRO and IMAS, respectively. Dr Craig Mundy was the liaison within the University of Tasmania and also acted as the Principle Investigator.

Detailed methods are provided in Appendix 3 (Stage 1; Objectives 1-3) and Appendix 4 (Stage 2; Objectives 4-6).

6.1 Stage 1 (Objectives 1 – 3)

Briefly, in Stage 1, information was obtained from fishery Management Plans, documents describing Australian, State-based abalone fishery management systems, state-based, fishery assessment reports, and peer-reviewed, published literature. Much of the information was difficult to obtain, especially for the abalone and other dive fisheries outside Australia. Three workshops were convened (Hobart, Tasmania; Mount Gambier, South Australia; Port Lincoln, South Australia). In each workshop participants (listed in Appendix 3) with experience in the application and interpretation of PIs at zone, state and regional scales formed an expert panel, typically comprising all stakeholders, ranked each PI as very useful (VU), useful (U), some use (SU), not useful (NU), and not applicable (NA). For numerical analysis, scores were assigned to each category (VU – 4, U – 3, SU – 2, NU – 1, and NA – 0). Panel members also identified additional PIs that were similarly ranked. A request for the same ranking was made to NSW, Victoria and WA via email (19 May 2010), but no feedback was received from these three States. Following review by the expert panel, the effectiveness of 11 PIs at detecting change in abalone abundance was evaluated at multiple spatial and temporal scales using three datasets: (1) a commercial greenlip fishery off Cowell in the Central Zone (CZ) of the South Australian Abalone Fishery (SAAF); (2) a fish-down (i.e. managed depletion of legal-size abalone, in excess of long-term sustainable limits) of blacklip in Waterloo Bay in the Western Zone (WZ) of the SAAF and; (3) an exploratory fishery for *H. roei* in the WZ of the SAAF. The three fisheries targeted different species under different fishing rules. Thus, each dataset is from discrete fishery, and shared few similarities.

6.2 Stage 2 (Objectives 4 – 6)

Stage 2 involved a wide range of different activities including:

1. Characterizing the fishery: a) describing the current management and its history and b) formally describing the behaviour of the abalone divers in the Tasmanian fishery.
2. Characterize spatial heterogeneity across numerous populations of biological properties: a) Growth; b) Size at Maturity; c) weight at length; d) emergence; and others.
3. Analytical exploration of current performance measures: a) the analysis of catch, effort, CPUE, and commercial catch at length data to explore the possible outcomes and

- potential contrast available in typical data available from the commercial fishery; and b) the comparison of alternative formal stock assessment models to suitable data from the fishery so as to consider model based performance measures.
4. Design and produce a management strategy evaluation (MSE) simulation framework for a viable abalone zone.
 5. Use the MSE framework to explore the trade-off between LML and TAC and also to test a number of alternative Harvest Control Rules relating to CPUE data.

While some of these sections were focussed on relatively simple analyses there were all aimed at enabling the development of the simulation framework to be used in the management strategy evaluation of management strategies in abalone fisheries. The methods for each of these components are given in detail in the methods sections from 15.4 onwards.

6.2.1 Characterizing the Fishery

6.2.1.1 The Current Management of the Fishery

A brief review was made of the array of current management measures and how they developed through time. Thus, the introduction of the legal minimum length (LML) and how that has changed through time was charted. Similarly the introduction of the quota management system is documented and described (see section 15.2, p.116). A knowledge of the current management is required if some notion of the underlying objectives for the fishery are to be understood. Unfortunately, these need to be inferred because in most abalone fisheries, certainly in the Tasmanian fishery, the particular objectives that the management aims to achieve are not stated explicitly and the aims that are stated remain so general that they do not materially assist with the yearly management of the fishery.

6.2.1.2 The Fleet Dynamics

The term fleet dynamics here refers to diver behaviour. This attempts to answer questions relating to how the fishery is sub-divided among the divers, whether the same divers are active each year, what is the spatial distribution of effort, how are catches distributed among divers, and whether some divers specialise in specific areas of the fishery or fish across broad geographic regions within the fishery? Underlying all of those questions is the general question: how do management decisions affect how divers distribute the fishing effort and catch and are there any unintended consequences from introducing those management decisions?

6.2.2 Characterizing Biological Properties

The abalone section of the former Tasmanian Fisheries and Aquaculture Institute, and now the Institute for Marine and Antarctic Studies, has, through the last 20 – 25 years, developed a database of biological observations on different abalone populations around Tasmania. This includes, as a minimum, tagging data on growth, biological observations on size at maturity, morphometric relationships regarding shape and weight, the size distribution of samples, and shell covering by epiphytes and encrusting organisms. The tagging data can be used to generate estimates of growth for 30 separate populations. The biological observations were used to produce weight at length relationships for 122 populations, size at maturity relationships for over 250 populations, and emergence parameters for over 40 populations (see section 19.4, p.209).

These fitted relationships and their related parameters were used to condition the Management Strategy Evaluation (MSE) simulation framework to have properties similar to

the east coast of Tasmania although the process is more general than that and given the data the model can be conditioned to be similar to any abalone fishing zone. The data requirements for statistically fitting the model to the east coast fishery are far too great to be feasible so the productivity of the simulated zone used in the MSE testing of alternative management strategies was plausible but arbitrarily selected.

6.2.3 Analytical Exploration of Current Performance Measures

6.2.3.1 Empirical Performance Measures

Data from the abalone catch and effort log books (originating from DPIPWE) and an IMAS database of commercial catch length distributions were used to search for trends in commercial catches, effort, CPUE, the spatial distribution of catches and the length frequencies of the commercial catch. The aim was to determine whether or not there were patterns that reflected changes in the stock status. If such patterns in these statistics exist then the possibility is raised of using the simple data as a useful performance measure for the fishery which could be used to assist with management decisions.

Catches and effort were considered and variations in the manner in which these may be reported were also examined. Out of the various data available the use of CPUE and of catch length frequencies is common in abalone assessments, so most attention was focussed on these.

The routines involved with standardizing CPUE data are described in section 16.2.3, and these are general and classical. There is a generally stated belief that CPUE data for abalone fisheries is invariably uninformative. Despite this CPUE data is used in numerous stock assessments for abalone, both formal assessments based on mathematical models of stock dynamics and more qualitative assessments. This question of the information content of CPUE is therefore pivotal.

If catch rates are informative about stock sizes and their dynamics in response to fishing then the expectation is that rising catches will lead to declines in catch rates and declining catches should enable catch rates to increase. This pattern appears to occur on both the east and the west of Tasmania, where reductions in catches were followed by increases in catch rates and increased catches also appear to have led to decreases in catch rates. If the time lag between these events is consistent through time, this would suggest both that there is a link between catches and subsequent catch rates and therefore that catch rates are reflecting the dynamics of the fished stock.

To test this relationship, linear regressions between catch rates and catches were carried out with sequentially increasing time lags by which the catch rates were pushed backwards. If there is a relationship between the two this relationship would be expected to be negative. This would imply that catches now should influence catch rates in the future, with high catches reducing future catch rates and relatively low catches allowing future catch rates to increase. By comparing the resulting correlations and the statistical significance of each relationship the optimal time lag can be determined (see section 17.4).

Analysing commercial length frequencies without an underlying formal model of stock dynamics is not so simple, however, it is possible to estimate the mean, median, and length of abalone at the 25% and 75% quartiles, and see how those might change through time. By using length frequency data from 2008 onwards the sample sizes are relatively

larger because data were collected in the processing sheds using data logging measuring boards. These estimates were compared with trends in the catch rates over the same time from the same areas from which the length frequencies were collected to determine whether they followed the same or a similar trajectory through time (see section 17.8, p.172).

6.2.3.2 *Model Based Performance Measures*

Two types of formal model were considered: relatively simple surplus production models and more complex size-based integrated assessments.

Surplus production models only require a time series of catches and of an index of relative abundance (usually CPUE). Despite this simplicity, if the fishery matches the assumptions of the analysis then the models can produce estimates of fishing mortality and of exploitable biomass. They do not account for size at maturity or related details so they do not relate to spawning biomass, but one assumption is that catch rates relate directly to exploitable biomass so they do provide information regarding that. Such models can be projected forward under different assumptions of catch (TAC) in a risk based context so as to produce estimates of the likelihood of different possible outcomes when a stock is fished in different ways. Details of the implementation and analyses involved with surplus production modelling are given in section 18.2 (p.178).

Size-based models require rather more data but the advantage of such models is that they can integrate an array of different data streams with each providing information about different aspects of the stock dynamics. Such models require information about various biological properties such as growth, size at maturity, and weight at length plus a time series of catches and of an index of relative abundance (CPUE again is most common). However, they can also include data about the length frequency of commercial catches, tagging information relating to survivorship or growth, population size distributions and any other data source that becomes available. Being more complex these models can also have more outputs, so they can generate fishing mortality rates, spawning and exploitable biomass estimates, predicted size distributions, and derived statistics such as unfished biomass and MSY. There are other advantages to such models in that they can be used to predict the effects of changes in the TAC as well as changes in the LML. Details of their implementation and related analyses are given in section 18.3 (p.186).

6.2.4 Design an MSE Framework suitable for Abalone

6.2.4.1 *Conditioning Abalone Simulation Models*

The ranges in the available growth estimates, the size at maturity estimates, the emergence estimates, and other factors, and any relationships between these different biological properties, were used in the characterization of the dynamics of the multiple populations making up the simulated zone. Probability distributions were placed around each biological properties and random values selected (in a manner that correlated with the other properties) for each population. The properties of the resulting zone emerged from the input variables being combined in the structure represented by the equations describing the model (see section 19.2, p.201).

6.2.4.2 *The Model Structure*

The dynamics of the simulations operate at an annual time scale and there is no distinction made between the sexes as they are deemed to grow in the same manner and are not distinguished by the divers. The size-structure that is used has 105 size classes of 2 mm from 2 – 210 mm, with the maximum size class acting as a plus group. This range covers the

expected sizes to be found in Australia. The 2 mm (equivalent to sizes 1-3mm inclusive) was selected because the size at which the first shell pore becomes defined is generally somewhere between 1-2mm (Prince *et al.*, 1988), and that is often deemed to be the start of the juvenile stage and to occur after two or three months (Cropp, 1989). The 210 mm plus group was selected because very few abalone, even in west coast Tasmania, grow larger than that. The 2mm size class was selected as a compromise between excessive computational load and the provision of fine detail for selectivity and growth.

While the sexes are combined there are, however, separate vectors of numbers-at-size for the cryptic and emergent components of the stock. The time step is annual with natural mortality being implemented in two halves with the remaining dynamics in between; details are given with the formal equations and the pseudo-code (see section 19.2, p.201). All recruitment is into the cryptic component and any fishing mortality is imposed on the emergent component.

Each simulated zone is generally made up of 70 separate populations (this number is chosen by the operator and could be larger or smaller) and each population is described using a collection of variables concerning its growth, reproduction, natural mortality, biomasses, numbers at size, and its fishery. Any detail required, such as the size distribution of the catch, the numbers or proportion remaining below the legal minimum length, the fishing mortality rate, the annual spawning biomass levels, or perhaps the state of depletion can be calculated and tabulated, plotted, or used in subsequent analyses. Generally, each zone was initiated to some pre-selected state of depletion level, then the dynamics are run for ten years at some selected constant initial TAC so that the variation inherent in any random elements can become fully expressed, and then whatever management strategy that is to be applied is imposed from year 11 out for another 40 years to year 50. From year 11 onwards the harvest control rule of the management strategy under investigation uses the values of the selected performance measures to decide on the future level of TAC which is fed back into the model and the cycle continues each year until the model reaches year 50.

Random variation is included in three places. The first and most obvious is with recruitment variation, the second relates to the data being used in the estimates of the performance measures used in the management strategies under test, finally the third is in the estimates of exploitable biomass available to the diver's (through diver knowledge of where they have previously fished). Recruitment variation includes both the variation around the spawning stock and subsequent recruitment relationship but also the occasional larger scale zone-wide recruitment events that occasionally happen, and the occasional relative failure of recruitment in individual populations. The details of recruitment dynamics remains a key uncertainty in abalone population biology. Even when the episodic events within populations only occur say one time in 25 years this can influence the outcome of an individual simulation and so this forms an important component of the variation seen in abalone stock dynamics. The variation in the data, usually CPUE, what makes up the performance measures is important because the usual assessments are only based on relatively noisy data. We do not have ideal data from the fishery (for example, different divers can record effort very differently so even if two divers were, in fact, identical in their catching ability they would record different CPUE). By including error this means that the harvest control rule used can never be perfect, but this will also reflect what happens in a real fishery more closely. Finally, The TAC decided for a simulated zone can be caught across that zone in different ways. The mechanism chosen to represent fishing behaviour is to use estimates of exploitable biomass in different reporting blocks (made up

of a number of the unit populations making up the zone) but those estimates have random noise added so that catches are not spread across different blocks in a perfect match with available biomass. The degree of variation added to this component reflects that observed in the characterization of the fleet dynamics.

An important aspect of including random variation is that it can be turned off to see how the system would operate if perfect knowledge was available. This is equivalent to recruitment being stable and a direct reflection of spawning biomass, that catch rates would be a perfect representation of the relative abundance of exploitable biomass, and that catches were distributed across a simulated zone in a perfect match with available biomass so that the proportion of the stock in each area that was removed by fishing was the same everywhere. While, of course, this is never expected to occur it identifies the system behaviour that random variation influences and obscures.

Once random variation is included in the dynamics then replicate runs can be made whose outcomes can be summarized in a variety of ways to illustrate the effects of different initial conditions and different ways of generating a management response to simulated fishery statistics produced each year from the model. This facility allows the investigation and testing of alternative arrangements.

6.2.5 Legal Minimum Lengths and Total Allowable Catches

In the management of abalone stocks around the south-east of Australia there is often a debate about at what size to set the legal minimum length (LML in Tasmania but other acronyms are used elsewhere; see section 15.4 on terminology). The argument is often about how raising a LML will lead to some areas being excluded from the fishery because the animals there do not grow large enough. This would mean that the amount of biomass that used to come from those excluded areas will now have to come from the remaining area which would imply that fishing mortality will increase. The MSE simulation framework was used to approach this issue by answering a number of questions. These were:

1. How many abalone are required for the same catch at different LML?
2. What effect does LML have on yield and proportion of mature biomass protected?
3. How are stock depletion levels affected by fishing the same TAC at different LML?

The methods relating to the simulation framework and its conditioning (see sections 19.1 and 20.3, p. 223). This work has already been presented to the 8th International Abalone Symposium held in May 2012, in Hobart, Australia and has also been formally published (Haddon and Helidoniotis, 2013). Detailed methods are also given in that paper, including a somewhat simplified version of the MSE simulation framework equations.

6.2.6 MSE Testing of Alternative Harvest Control Rules

Stage 1 of the research presented in this report focussed on trying to identify alternative possible performance measures (PMs) that might be of use in managing abalone fisheries. However, before launching into a detailed examination of PMs different to those currently used, it is worthwhile examining the behaviour of PMs currently used, such as catches, catch rates, and length frequency distributions of landed catches. These PMs have been instrumental in helping to maintain various relatively large abalone fisheries in Australia for 50 years or more (Mayfield *et al*, 2012). Perhaps more important than the PMs used are the Harvest Control Rules or set of Decision Rules (HCR) in which the PMs are used.

Assuming the same quality data is available, the very best PM possible will fail to provide good management advice if embedded within a bad HCR or a poor set of Decision Rules. It appears to be a reasonable strategy to first optimize the use of PMs currently used to manage Australian abalone fisheries by examining how they interact with different HCR and only then add different PMs and HCRs; this is the strategy adopted here.

The two alternative families of HCR that were considered were chosen to attempt to reflect current informal usage. These included HCRs that used the gradient of recent catch rates from a fishery as a basis for calculating a current TAC modifier; these have the basic idea that if the gradient of recent CPUE is positive then catches can also increase, but if the gradient is negative then catches should also decrease. Exactly how that is implemented can vary but the underlying principle is clear (see section 21.2.2, p.240). The alternative HCR considered was based around the notion of setting a target CPUE so that if the recent CPUE was below the target then catches would be reduced while if it were above the target they would be increased; with the degree of change in TAC usually reflecting the deviation from the target. In the case examined here the difference between the current CPUE and the target were put into a Decision Criterion Framework, which ranked the degree of deviation and proposed a positive or negative proportion change in the TAC as appropriate (see section 21.2.3, p.241).

Very often there are modifiers to any set of management rules and one that was applied to both the CPUE gradient HCR and target CPUE HCR was to only allow the TAC to decrease down to half the initial TAC (see section 21.2.4).

In all cases a simulated zone that was conditioned to be similar to the Tasmanian east coast was used. This meant that it had an average biological minimum length (size at maturity plus two year's growth) of about 138 mm, so the two year rule was best matched by a LML of 138mm. The maximum sustainable yield of the simulated zone was about 630 t. It must be remembered that the simulation is not "fitted" to the east coast, it merely has properties similar to the eastern zone. The actual productivity of the eastern zone remains unknown.

To test the relative performance of the two alternative HCR, 27 combinations of three LML (127, 132, and 138mm), three initial depletion levels ($30\%B_0$, $40\% B_0$, and $50\% B_0$), and three TACs (450, 600, and 800 tonnes) were used as starting conditions for simulations. These options were doubled to 54 for each HCR by including or not the use of a TAC minimum of half the initial TAC (see section 21.3, p.244).

The test simulations included single runs, with and without random variation, as well as replicate runs of each combination. 100 replicates were run in each case, with each replicate involving one zone made up of 10 blocks (statistical assessment units) made up of 70 populations, run for 50 years, with the first ten years at the constant initial TAC as a way of allowing any variation present to become fully expressed in the simulations.

7 Results and Discussion

7.1 Stage 1 (Objectives 1 – 3)

Management objectives among the Australian state-based abalone fisheries encompassed biological (includes ecological and environmental), economic, governance (management), and social categories, with the biological objectives providing the principal management direction. There were substantial differences among states regarding the specificity, diversity and number of abalone fishery management objectives, and many were general in nature and, at times, contradictory. Nevertheless, as all states prescribe biological and economic objectives, these reflect the (1) importance of ensuring the stocks are fished sustainably and within an ESD framework, and (2) high value of the product, licences and quota. The biological objectives exhibited the highest level of consistency among States.

A diverse range of PIs are prescribed for the assessment of Australian state-based abalone fisheries (Appendix 3). Overwhelmingly, most PIs relate to assessing fishery performance against biological objectives and, almost exclusively, those relating to sustainability rather than ecosystem integrity. The PIs are obtained from a broad range of sources including fishery-dependent and fishery-independent data and outputs from numerical models, with catch rates being one of the most common PIs by which Australian state-based abalone fisheries are assessed against biological objectives (Appendix 3).

As fishery assessment evolves through an ongoing process of continual improvement, studies that provide more accurate, more precise and less costly approaches to assessing fishery performance, for the purposes of making management decisions can yield potential new PIs. Several of these were identified in this study, including spatial indices of stock status, weight-grade data, a direct measure of potential egg production and estimates of maximum sustainable yield (B_{MSY}) and maximum economic yield (B_{MEY}) from models (Appendix 3). In addition, perhaps one of the greatest opportunities for developing novel PIs is to consider those based directly on industry knowledge and perception. This would provide one formal mechanism for incorporating ‘diver assessments of stock status’ into determining harvest strategies and harvest control rules. Whilst most fisheries currently use this information in conjunction with scientific assessments to determine TACCs, existing mechanisms tend to be ad hoc and informal. This approach may be most beneficial for areas of the fishery that do not support large catches and/or where information to assess fishery performance is more limited.

Despite Haliotid fisheries outside Australia being both extensive and widespread, none of these had or have management systems incorporating harvest strategies based on PIs with associated reference and trigger points (Appendix 3). However, the Californian abalone fishery in the USA has an Abalone Recovery and Management Plan and a draft harvest strategy has been developed for the NZ abalone fishery (Hills 2009) and these are detailed in Appendix 3. We also examined objectives and PIs for other dive fisheries. Like the non-Australian abalone fisheries, information on these was limited (Appendix 3) and much of the available documentation lacked clear management objectives, stipulated PIs and associated harvest strategies – despite formal fishery management plans for these fisheries being more common (Appendix 3). Thus, reviews of these two fisheries groups yielded little value.

Selecting appropriate, robust PIs, which accurately measure changes in fishery performance, and are thus suitable for assessing abalone fisheries against the specified biological, economic, social and governance objectives is challenging, with the lack of conformity among States reflecting the difficulty of identifying appropriate PIs, the operational and both legislative and management differences among fisheries. In addition, the suitability of the broad range of PIs identified in this study to act either as an ‘early warning signal’ or as an indicator of improving resource status is poorly understood. As management decisions for these fisheries are collectively informed by the PIs, future sustainability of these fisheries requires that these PIs be informative. Thus, these PIs must provide clear, timely indications of variation in abalone abundance and/or population structure, and be sensitive to, and effective at, detecting biologically-meaningful changes. In the absence of this level of sensitivity, these PIs will fail to identify Zones/Regions/Reefs where the resource could sustain additional fishing pressure, or may be overfished.

Uncertainty in the quality of the PIs has limited development of formal, harvest-control rules that prescribe clearly-defined management outcomes and have also resulted in assessments of stock status being only loosely based around the prescribed PIs, with deviations from the prescribed assessments leading to the development and use of ‘informal’ PIs. This move has been required because the need for assessing stock status has not diminished, despite the absence of a reliable and accepted set of PIs. Consequently, this study aimed to determine those PIs that are likely to be most informative, and consequently most suited, for use in assessment of Australian abalone fisheries. This process was undertaken through a number of steps. First, expert panels evaluated this list of PIs generated by the national and international review. Then, those PIs with the most potential were evaluated against known fishery performance. Finally, a management strategy evaluation provided a more thorough and formal assessment.

Expert panel members identified an additional 18 potential PIs and, thus, a total of 75 PIs were considered at each of the three workshops (Appendix 3). Three PIs – raw CPUE ($\text{kg}\cdot\text{hr}^{-1}$), proportions of large and small length classes in the commercial catch and diver assessment of stock status – were ranked as VU at each of the three workshops. In contrast, four PIs were ranked as NU by each expert panel. These PIs were catch ratios between species, CPUE ($\text{kg}\cdot\text{month}^{-1}$), $F\cdot\text{yr}^{-1}$ and the number of prosecutions as a measure of illegal catch. Apart from these seven PIs, there was generally little consistency in the outcomes from the three workshops (Appendix 3). However, the percentage of maximum score, determined by scoring the ranks provided to each PI at each workshop, provided an objective approach to synthesising the data. For the biological PIs, 23 were $\geq 75\%$ of the maximum score (Appendix 3). Of these, there were three at 100%, two $>90\%$ and a further six at $>85\%$. The lowest value, 25%, was attributable to four PI. Whilst fewer economic PIs were ranked, one – GVP – had a value of 100% and a further eight were $>85\%$ (Appendix 3).

The expert-panel workshops undertaken in this study provided the first step to assessing the ability of the PIs used in the management of abalone fisheries to act as an ‘early warning signal’ of decline, as an indicator of improving resource status, or as an index of sustainability for these fisheries. Such an approach is not unique (Sammarco 2008, Southall *et al.* 2009, Zajicek *et al.* 2009) and, because the expert panels were typically comprised of all stakeholders (i.e. divers, licence holders, research scientists and fishery managers), there was a balanced, broad representation which contributed substantial experience to the assessment. The process of assessment was also rapid, thereby facilitating a timely con-

sideration of the PIs identified. However, expert-panel approaches are entirely qualitative and “opinion driven”, can fail to consider the “achievability” or quantitative elements of some PIs, and may be biased. Thus, individual PIs can receive a ranking (i.e. high or low) that is inconsistent with their historical or potential performance.

Determining an overall score for each PI provided a mechanism to synthesise the rankings of the three workshops to a single value. For the biological PIs, 23 achieved a rating of $\geq 75\%$ of the maximum possible score and, consequently, can be considered “preferred” PIs (Appendix 3). This approach was also useful for identifying “preferred” economic PIs. GVP, and a further eight economic PIs, had a rating $>80\%$, which distinguished them from the others as “preferred” (Appendix 3).

Overall, the expert panel approach enabled the perceived relative value of 75 potential abalone PIs to be identified by stakeholders in the SA and Tasmanian abalone fisheries. This rapid, but qualitative and opinion-driven process provided a “first cut” analysis, and enabled these PIs to be broadly categorised as “preferred”, “non-preferred” and “State/zone-specific”. This information is useful for selecting suites of PIs suitable for assessment of abalone fisheries in each State or zone. Importantly, the suite of PIs selected should be as diverse as possible, and avoid autocorrelations among PIs (i.e. each PI should measure performance of a unique component of the fishery). This will require consideration of existing management and legislative arrangements, research capacity to service assessment of fishery performance against the PIs and the availability of data.

There was little consistency in PI trends among fisheries but numerous PIs changed through time in a manner consistent with declines in fishery performance and reflective of reductions in legal-size abalone abundance. These changes were apparent in three ways. First, PI estimates changed substantially through time. This was particularly evident for PIs at Cowell and for mean size, median size, proportion small and proportion large in Waterloo Bay (Appendix 3). Second, estimates of 12 PIs changed significantly through time with 10 of these 12 consistent with decreases in abalone abundance. Third, PIs derived from commercial-catch-sampling data (i.e. mean size, median size, proportion small and proportion large) displayed differences through time that were more consistent with decreases in fishery performance and reductions in legal-size abundance than those from CPUE and mean daily catch.

Although data from three fisheries were used to evaluate 11 candidate PIs, the data for Cowell were the only data available from a fishery operating commercially that had a consistent number of experienced fishers targeting a familiar species over an extended period of time (4 years). This fishery was initiated to allow harvest of a previously underexploited greenlip population, but catches were not sustained and the fishery was closed after four years of fishing. Thus, the data for Cowell are most similar to those that would be expected from a fishery experiencing a rapid decline in stock levels. In contrast, Waterloo Bay comprised a fish-down to deplete stock levels and the Roei fishery was designed to investigate the potential of a commercial fishery on this species. Thus, the primary purpose of data collection for all these fisheries was not to test the performance of prospective PIs. Nevertheless, these data have been used to provide the preliminary analyses of PIs undertaken.

Initial, quantitative application of the 11 PIs analysed highlighted marked changes in some measures of fishery performance. Notably, at Cowell, several PIs displayed signifi-

cant decreases through time. There were also prominent (but not significant) decreases in median shell size and the proportion of large shells, and an increase in the proportion of small shells in the commercial catch. Similarly, in Waterloo Bay, there were changes in the size structure of the commercial catch, as mean and median size and the proportion of small and large blacklip harvested varied through time, but yielded substantial contrast in their initial and final values, consistent with expected declines in abundance.

Here, data availability and sample size was an issue that frequently restricted the power of the analyses. For example, in Waterloo Bay, fishers stopped fishing in the fish-down when their own CPUE approached levels below those achievable elsewhere in the fishery and at which they would not normally fish, despite prior agreement to meet the objectives of the fish-down. This meant that the number of fishers operating at the spatial and temporal scales assessed was lower than anticipated and there were fewer data representative of divers and diver-days. This likely had a greater effect on the assessment than would otherwise be the case in a 'normal' fishery where time and space constraints are absent. Some data were also biased. For example, again from Waterloo Bay, poor weather conditions during the first 10 days of the fish-down limited fisher access to abalone in one experimental area and kept initial levels of CPUE and MDC low. These issues highlight the need for explicit consideration and evaluation of data available for assessment (Chen *et al.* 2003) and the need to consider decision rules that exclude data biasing measures of fishery.

Nevertheless, the preliminary analyses of candidate PIs highlighted the strength of those based on commercial-catch-sampling data, along with the need to explicitly consider (1) the suite of PIs that most closely match management objectives; (2) sensitivity of PIs to detect change, particularly their ability to measure decreases in abundance prior to stock collapse; (3) minimum data requirements; (4) factors that bias data; and (5) statistical methods employed.

7.2 Stage 2 (Objectives 4 – 6)

7.3 Characterizing the Fishery

7.3.1 The Current Management of the Fishery

The current management of the fishery (see section 15.2, p.116) does more than simply describe the history of the fishery, it also describes the variation through time that has been experienced. In addition, it becomes clear that even though there have been different approaches to management in the different jurisdictions they all share common elements. Mayfield et al, (2012) provide a more detailed description of the Australian Fisheries and, in combination with the more detailed description of the history of the Tasmanian fishery, it becomes clear that the most important thing missing from all of the fisheries is an explicit objective towards which the management of the fishery is aimed. South Australia has recently adopted a harvest strategy and Tasmania is in the process of testing how best to do the same in Tasmania. However, to date, these are only going to be using empirical performance measures and relatively simple harvest control rules. What this means is that, while the harvest strategy implies an underlying objective (achieve and acceptable status and operate to remain there), there will be no understanding of the underlying dynamics. Understanding the dynamics is not essential to gain successful management in terms of sustainability and profitability, but presently there are still no explicit management targets to aim towards, so it is unknown whether they would be optimum or not. The South Australian harvest strategy has yet to be tested but may be one that attempts to maintain the status quo. The current relatively informal and qualitative TAC setting process in Tasmania is not governed by any explicit objective but one can infer that the aim is to maximize catches while trying to maintain catch rates.

After more results from the simulation studies have been described it will be useful to revisit this issue of what constitutes a suitable objective.

7.3.2 The Fleet Dynamics

Fleet dynamics is represented by diver behaviour in abalone fisheries. It can be characterized in a number of ways including how divers distribute their effort and catches spatially (see **Figure 30**, p.135, **Figure 31**, and **Figure 36**), and how catches from particular areas are distributed among divers (see, for example, **Figure 32**, p. 137, and **Figure 38** to **Figure 40**). The first question about the spatial distribution of effort and catch relates both to the distribution of the stock (larger amounts of available biomass will generally entail more effort and catch; although not always) and the propensity of each diver to be a spatial specialist or generalist. The second question about how catches are apportioned among divers relates to which among the available dive licences can be classed as active divers and would be important in decisions concerning the total catch share required by individuals to be able to make a full time living from the industry as a diver. Exactly what constitutes the optimum number of divers within the various fisheries is a very difficult question to answer because neither economic nor social objectives relating to the divers are defined. Whatever objective is adopted will almost automatically assist in the definition of what constitutes optimum.

The role of this current work is not to suggest management policy but rather to provide information that may have value to making such decisions. Thus, the primary aim considered when trying to characterize the fleet (or diver) behaviour within abalone fisheries

was to determine whether diver behaviour (in terms of the distribution of catch spatially and the distribution of catches among divers) changed in the face of management changes.

Details are only given concerning the Tasmanian abalone fishery because that was the only one for which all fisheries data was available. In fact, there have been a range of diver responses to changes in management. Some large changes have occurred in the management of blacklip abalone in Tasmania since 1985 (see **Figure 23**, p. 120). These include a large reduction in TAC following on from 1985, relatively large changes in the LML, differently in the east and west, and the introduction of zonation in the fishery in the year 2000. All of these changes can be expected to have had influences on the fishery, especially the introduction of zonation and changes in the TAC. Changes brought on by altering the LML may be harder to detect.

7.3.2.1 *Changes Prompted by Altering TACs*

In 1985, a quota management system was introduced in Tasmania in recognition that stocks were in a seriously depleted state and that catches were too high. With the introduction of quotas reported catches declined from about 4163 t in 1984 to 2075 t in 1989, settling at 2100 t in 1991 until the end of 1996. This had the effect of shifting effort and catch to focus mainly on the east coast, which is very much easier to access and to fish in most weathers (see **Figure 33**, p. 138, **Figure 34**, and **Figure 35**). Fortunately, there appears to have been some successful recruitment events because stocks on both east and west coasts began to recover relatively quickly, with the east coast recovering earlier than the west despite the increased catches. This eventually led to an increase in the overall TAC to 2,520 t (which included ~200 t of greenlip catch) in 1997. The recovery of the stock is reflected not just in the base catch rates but in a change in the relationship between effort and the resulting catch (**Figure 1**). Assuming that 1995 is a transition year between the two states of the fishery then there are a linear relationships between effort and catch between 1985 and 1995 and between 1995 – 2012. The gradients of the two regression lines for these two sets of years is not significantly different (**Figure 1**) but there is a 533 t different in the intercepts. Exactly what brought about this rapid increase in yield for the same amount of effort is not known but between about 1993 – 1998 there is an obvious transition between the divers who dominate the fishery in terms of catch (see **Figure 38**, p. 141, **Figure 39**, and **Figure 40**). This is not so much a change in behaviour as a change in which divers are fishing. Whether this was in response to the original change in TAC or because of other reasons is unknown but it certainly constitutes one of the major changes in the fishery.

It might be thought that the apparent change in catchability is simply due to increased effort in the west where catch rates tend to be higher (**Figure 2**). However, while the separation of the two time periods is clearer on the west, there remains a change in the character of the relationship between effort and catch even though it differs on the two coasts.

In addition to the changes in who was actually doing the fishing, the proportional distribution of catches also altered (**Figure 3**). While there have been changes in the total number of divers reporting catches through time these changes have been relatively minor. However, following the TAC changes in the late 1980s there were large changes in the number of individual divers reporting more than 20 t (**Figure 3**).

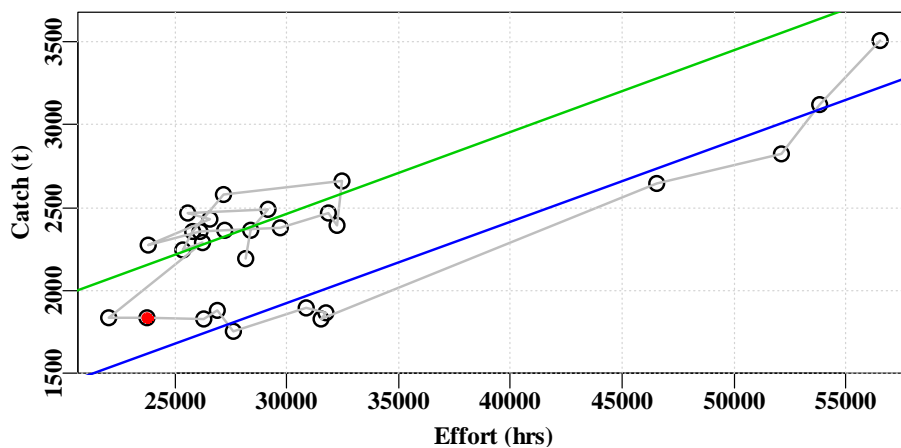


Figure 1. The relationship between reported effort and reported catch from 1985 (top right point) and 2012. The red filled point at bottom left is 1995, a transition point between different states of the fishery, the TAC was increased from 2100 t to 2520 t in 1997. The lower, blue line is a linear regression between catch and effort from 1985 – 1995 giving $\text{Catch} = 0.0489 \text{ Effort} + 458.988$, while the regression from 1995 – 2012 was $\text{Catch} = 0.0493 \text{ Effort} + 985.877$.

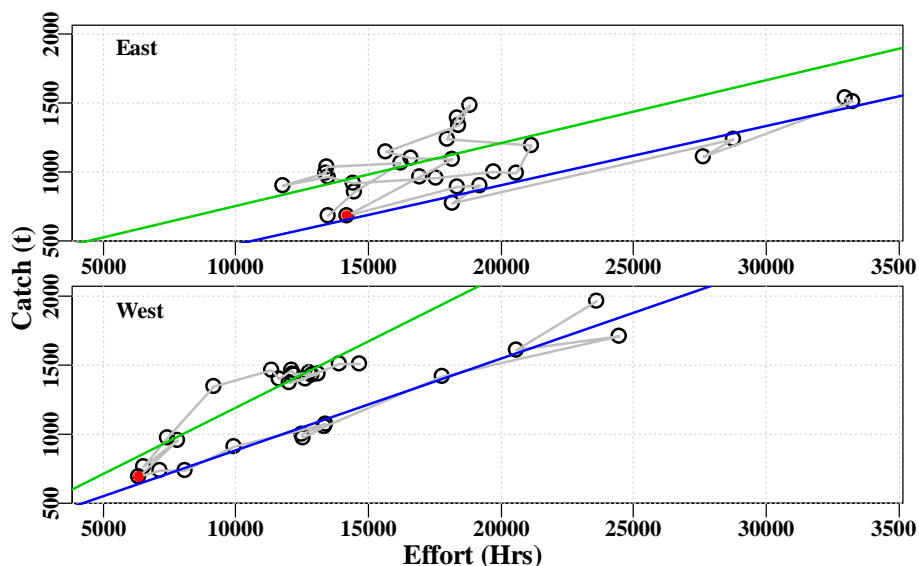


Figure 2. The relationships between total catch and total effort for the east and west coasts of Tasmania (split on 146.5°). In the east the transition year was 1992 while in the west it was 1996, which was also the shared year in the respective regressions relating to the blue and green lines. It is noteworthy that the final year, 2012, on the east exhibits a lower point than 1992 in the total statistics; this confirms that the east coast is currently in a relatively poor state.

The recovery of the stock can be seen in the number of divers that began to catch more than 20, 30, and even 40 t through the 1990s. The numbers catching these larger amounts began to become relatively stable after about 1998, although there were some changes in the total number of divers. This is a different way of looking at the changeover in the dominant divers within the Tasmania fishery.

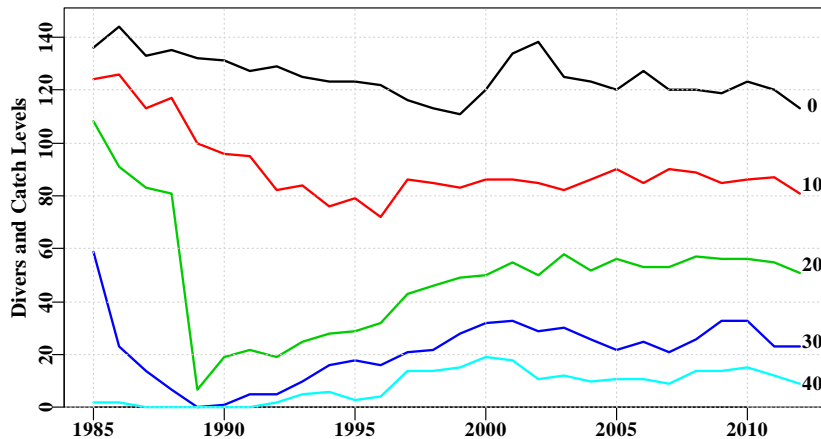


Figure 3. The number of separate divers reporting catches through time. These can be greater than 125 as more than one diver can use the same licence at different times in each year. The five different lines relate to different catch levels, thus the 0 line relates to any diver that reports a catch, the 10 line relates to those divers whose total annual catch is greater than 10 t, the 20 line is for divers reporting annual catches greater than 20 t and so on.

7.3.2.2 *Changes Following Alterations in the LML*

There may have been changes in diver behaviour following changes in the LML but there are generally obscured in the changes that followed on from the TAC changes. Very often there have been changes in both things close together and this certainly makes it difficult to separate out their relative effects.

7.3.2.3 *Changes Following the Introduction of Zonation*

The biggest change in the structure of the fishery occurred in 2000 with the splitting of the east coast from the west coast and the introduction of separate TACs for each. This action was decided upon because only about 450 – 600 tonnes were being caught in the west while 1400 – 1600 t were being caught in the east. Zonation was aimed at forcing a more even distribution of catch across the entire reach of the fishery. There appears to have been a greater effect on the east coast than the west, with a more even distribution of the available catch among the active divers (see **Figure 38**, **Figure 39**, and **Figure 40**). This also occurred in the southern blocks of the western zone (blocks 9 – 12) but in the north the effect appears to be rather less marked (see **Figure 41**). In this case, no effects can be seen in the number of divers catching different amounts (see **Figure 3**), although following the introduction of zonation there was a relatively large increase in the total number of divers reporting catches but in terms of the most effective divers, the numbers that were reporting catches greater than 10 tonnes remained remarkably stable right through zonation.

7.3.3 Non-Management Changes of Significance

Besides changes in management other influences can change diver behaviour (**Figure 4**). About the time that the TAC first increased in 1997, following the reductions in the late 1980s the live export trade began to increase as a proportion of the total blacklip landings in Tasmania. Apart from the relative ease of fishing in the east relative to the west, this is likely to have been an additional driver for divers to fish on the east coast. East coast abalone, at the time, were more suited to the live trade than the larger animals from the west coast. The live trade became such a dominant factor because the beach value of live mar-

ket abalone was higher than abalone destined for canning or other processing methods. With the maximum beach price either remaining stable or declining in real value through the 2000s (Tarbath and Gardner, 2012) this increase in value constituted a real incentive to fish for the live market and this generally limited to locations that were suitable for that purpose.

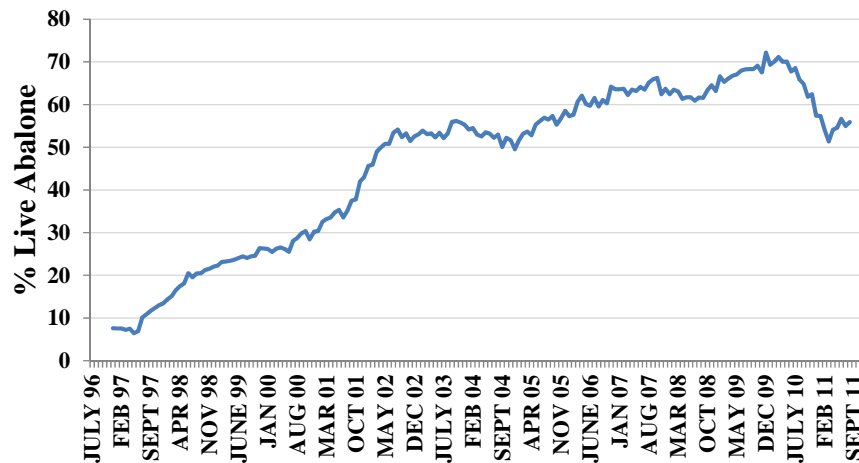


Figure 4. The proportion of landings from the whole blacklip fishery that were destined for the live market (data courtesy of Tony Johnston, Tasmanian Seafoods).

Across the fishery the number of divers who report landing different amounts of abalone has remained relatively constant over the last 10 years, although the number of divers reporting relatively small annual catches less than 10 t is declining slowly. There has been increasing concern over the difficulty in making a living when diving for abalone but this would appear to be more a function of a failure to maintain or increase the real value of the beach price than the amounts that individuals are catching, although it is possible that the same number but different divers are doing the significant catches each year, which is something that could be further investigated. One solution would be to reduce the number of divers and generally increase the amount caught by individuals, another would be to develop stronger markets willing to pay more, in real terms, for the product.

7.3.4 Characterizing Biological Properties

The abalone section of the former Tasmanian Fisheries and Aquaculture Institute, and now the Institute for Marine and Antarctic Studies, has, through the last 20 years, developed a database of observations on different abalone populations around Tasmania. However, through an initiative put together by Dr Craig Mundy, the most recent Abalone Section leader, it was only by about 2004/5 that this database was redesigned so as to allow access to its contents to become relatively simple and general.

It is well understood that, in abalone, there is a great deal of spatial variation in the biological attributes relating to growth, morphology, and maturity. Many studies have confirmed this (Worthington et al., 1998; etc), in Tasmania while there are numerous samples the full extent of variation is still to be documented (although variation in size at maturity is a partial exception (Tarbath *et al*, 2001). Here, for as many separate populations as were available in the database, we analysed relationships relating the total weight to length, the ma-

turity vs length and size at maturity relationships, the size at emergence, and produced growth curves; these analyses were for populations right around the coast of Tasmania (see section 19.4). This had not been done before at this scale and was required to permit the conditioning of the underlying dynamics of the MSE simulation framework so that it could appear to be similar in its properties to a selected zone. The growth of abalone is highly variable around Tasmania and is an important component of size-based models. Maturity is obviously vital in the estimation of the spawning biomass in any population and so how it varies is also important, and the weight at length is important for converting numbers at length into biomass estimates.

7.3.4.1 *Length to Weight Relationship*

As is typical for most marine creatures there was a power relationship between weight and length so that weight tended to increase in proportion to the volume of each creature (approximated by the length cubed: aL^3 ; section 19.4.2, p.209, and **Figure 74**).

Fortunately for the MSE framework a simple relationship was found to exist between the two parameters making up the 122 relationships for populations around Tasmania. This meant that by randomly selecting one value from its observed distribution it was possible to estimate the other parameter given this new relationship (see **Figure 75**, p. 210 and **Figure 76**).

7.3.4.2 *Size at Maturity*

There were data from a very large array of populations relating to the size at maturity but there was also a wide degree of variability the full range of which spanned values from about 70 – 132 mm for the size at maturity; defined as the length at which 50% of a population would be expected to be mature (see 19.4.3, p.211; **Figure 78** and **Figure 79**)

7.3.4.3 *Growth Parameters*

Any simulation model to be used for testing abalone harvest strategies needs to be able to represent the stock dynamics in both a plausible and defensible manner. The construction of the simulation framework therefore required a number of preparatory steps that facilitated generating the required plausible description. First the formal selection of appropriate mathematical sub-models to represent the various biological properties needed to be reviewed. This was straightforward for properties such as the size at emergence and the size at maturity because a classical sigmoidal curve generally provides the most acceptable representation of events. However, during and following the original development of this project (in 2006/2007) the review of growth led to a rejection of the previously used von Bertalanffy curve and a whole new growth model for abalone being developed. In addition, its validity was tested by determining whether it was preferable to classical curves used preciously and elsewhere (the von Bertalanffy and the Gompertz curves). These developments continued through into the initial years of the formal project once it finally began (Haddon et al, 2008; Helidoniotis et al, 2011; Helidoniotis and Haddon, 2012, 2013). A strong emphasis was given to characterizing growth in abalone because of its powerful influence over the population dynamics and productivity.

The inverse logistic growth model now used with abalone, at least in Tasmania (although its use elsewhere is increasing) has more parameters than the classical von Bertalanffy curve. Relationships were examined between the various parameters obtained from the 27

populations from around Tasmania that were available when these analyses were conducted (see **Figure 80** and **Figure 81**; **Table 29**; section 19.4.4, p.214). Remarkably, the 27 sets of parameters were found to lie scattered about on a three-dimensional flat surface, indicating the strong collections between the parameters (see **Table 31** and **Figure 81**).

7.3.4.4 *Spatial Heterogeneity of Biological Populations*

The basic assumption underlying all of these analyses relating to the characterizing the biological properties of the various populations sampled is that these populations are expected to vary in their properties spatially. Abalone are notorious for varying biologically on very small spatial scales. One of the aims of the management strategy evaluation is to investigate the role of uncertainty in our assessments and how it influences the performance measures and harvest control rules that we might use to manage each fishery.

The assumption that abalone are as variable as thought is based on numerous samples taken in numerous studies. However, it should be considered that sampling abalone is not a simple undertaking. Obtaining a representative sample of the full size range of a population is often very difficult. Helidoniotis and Haddon (2013) describe a study where simulated ideal growth related tagging data is randomly sampled to generate what appear to be typical data taken in wild abalone samples. When these samples have different growth curves fitted to them each type performs with more or less ability to reflect the original curve which was used, with random noise, to generate the sampling data. The variation possible from such random sampling was large and encompassed published ranges for different populations, even populations in different jurisdictions.

While this work (Helidoniotis and Haddon, 2013) is only a first consideration of this issue, it is a prime candidate for further work. The scale of real spatial heterogeneity is one of the factors making the validity of applying formal stock assessments methods appear questionable when applied over usefully large scales. Large scale variation is not surprising when there is a wide range of different habitats and conditions under which abalone populations are living, but, for example, there are stretches of the Tasmanian west coast which appear to experience remarkably similar physical conditions and yet variation is still perceptible. More work is required in relation to this issue.

7.4 Analytical Exploration of Current Performance Measures

7.4.1 Introduction

Performance measures (PMs) can relate to numerous variables concerning the fishery and the stock being fished. There are two major forms of PM, those termed empirical and those termed analytical. Where a PM is based directly on data collected from the fishery these would be empirical PMs and where such data is further analyzed, perhaps derived from a formal model, and the derived statistics used as the PMs, these would be analytical PMs. Empirical PMs include such things as catch-per-unit-effort (CPUE) or the proportional distribution of catches across different areas within a zone, while analytical PMs include model derived statistics such as stock spawning biomass or fishing mortality rate estimates. Many more assumptions are required when using analytical PMs, however, because many of these are taken to relate directly to the fished stock's dynamics through time they are often afforded a higher value in the interpretation of a stock's status. Which type of PM is used in a particular situation is directly a function of what harvest control rules (HCRs) are in place and, more fundamentally, towards what objectives the given

fishery is being managed. If the HCR adopted relates directly to spawning biomass then empirical PMs are not capable of addressing such requirements, although they can be used if treated as acceptable proxies for stock biomass (Haddon, 2012).

If multiple empirical PMs are used in combination (for example trends in catch rates and trends in the commercial length frequencies) this should be more informative than single empirical PMs. Using empirical PMs, and possibly multiple PMs, is an option when there is insufficient data available to fit a fully operational integrated analysis; in effect the integration of such disparate data streams is being done qualitatively rather than formally and quantitatively.

7.4.2 Empirical Performance Measures

7.4.2.1 *Catches*

Catches by themselves provide little information regarding their sustainability through time (Hilborn and Branch, 2013), although if a long time series is available and show no sign of diminishing this would constitute at least circumstantial evidence that an area can be productive at the observed levels (see section 17.2). However, having said that, in section 20.4.5 (p.229), when considering the changes in the numbers of abalone required to catch a TAC, long term simulations of fishing a zone demonstrated that some zones can appear to be sustainably productive over a 50 year period while depleting slowly. This is possibly a reflection of the flatness of the productivity or yield curve for each zone (see section 19.6, p.218; **Figure 83**) implying that catches greater than the maximum yield can be sustained for quite a long time as spawning biomass declines slowly. Thus, catches by themselves are not an adequate PM of a stock's status or productivity. However, if combined with such things as observed length frequencies in the catch or with catch rates, then possibly these together can indicate whether a particular catch level is leading to declines in the resident stock. However, that starts to be attempting to approximate the intent of a formal stock assessment model that integrates such data streams into a coherent whole (if it can).

7.4.2.2 *Effort*

Like catch, the effort applied through time within a zone only allows very crude comparisons between years, although this may have direct interest when considering economic inputs and possible related PMs (see section 17.3, p.148). Fishing effort needs to be combined with other data streams to become informative about the stock status. Those other data streams can include catch, location, and time of year. When combined with any of these things (or other factors) then effort can become very informative concerning the dynamics of a given fishery and possibly of the stock being fished.

Defining effort is another issue when it involves a diver fishery. Ideally effort would be characterized as the time spent underwater searching for abalone. This ignores the time it takes to travel to a location, which may have importance when divers decide where to fish; but that deals with fishing decisions rather than stock abundance. However, at what level the available information should be integrated has never been considered in detail. Should effort be defined in hours per dive, or should it be hours per day, or what level of summary should be used to best separate noise from information? There is a suggestion to use as a PM the number of short dives where the diver decides it is not worth while fishing in a spot despite prior expectation. This will be explored once the GPS data logger data builds up a time series across years. The mixture of searching and fishing complicates

the interpretation of effort and catch rates. With the advent of depth data loggers and the complete coverage of the Tasmanian fishery the opportunity to explore this question is becoming available, but insufficient data have been collected to date to work on that here.

7.4.2.3 CPUE

An index of relative abundance is commonly used in the monitoring of wild populations and when assessing the status of any fished stock. It is certainly often used in the abalone fisheries in the south-east of Australia and so this was given a detailed treatment (see section 17.4, p.148).

In fisheries, estimates of catch per unit effort (CPUE) are often used in assessments (Hilborn and Walters, 1992; Punt *et al.*, 2001; Little *et al.* 2011), however, the validity of this use depends on changes in CPUE being proportional in some way to changes in abundance. The underlying assumption behind this use of CPUE is that there is a simple relationship between the estimated catch rate and the amount of exploitable biomass available each year.

Unfortunately, there are many circumstances where CPUE is not linearly proportional to abundance. Harley *et al* (2001) analyzed over 200 scalefish datasets where CPUE estimates could be compared with fishery independent abundance surveys. The majority of the fisheries they considered exhibited some degree of hyper-stability, which implies that CPUE does not decline at the same rate as the stock abundance. If such hyper-stability is unknowingly present this could bias any management advice based on using CPUE as an abundance index. Hyper-stability can arise where catch rates exhibited by a group of fishers can be modified by the fishers changing their behaviour or fishing patterns. Classic examples of this can be found in fisheries involving hand collection, such as with abalone species, where divers collect the animals individually from the sea bed. In some circumstances divers can change their behaviour in order to maintain catch rates despite a decline in abalone availability, thus CPUE becomes hyper-stable and hence less informative about abundance. In New Zealand, in some formal size-based assessment models (e.g. Breen *et al.*, 2003) hyperstability has been omitted and in other instances (Breen & Kim, 2005) non-linearity has been included.

Despite the paradigm that commercial catch rates are unreliable and uninformative (Prince and Hilborn, 1998), all formal abalone stock assessments in Australia and New Zealand include the option of using commercial catch rates as an index of relative abundance, and often rely greatly on such time series (Breen and Smith, 2008; Breen *et al.*, 2003; Gorfine *et al.*, 2005; McKenzie and Smith, 2009; Worthington *et al.*, 1998). Less formal, empirically based fishery assessments in Tasmania and South Australia also use abalone catch rates to inform management (Burch *et al*, 2011).

As well as identifying hyper-stable catch rates as a problem for abalone fisheries, Sloan and Breen (1988) also pointed out that abalone catch rates are made variable by divers having different reporting practices for effort (e.g. effort as hours on the water or sometimes hours underwater). This source of variation remains a problem but many factors other than the stock abundance can also influence the apparent CPUE (for example, the diver doing the fishing, the month of fishing, the location of fishing, etc). The standard approach used to account for the effects of such factors is to apply statistical standardization to the CPUE data. While this ought to be a significant improvement on using the raw catch rates, it is the case that standardizations that use diver as a factor would fail to ac-

count for the situation where some of the divers altered their usual behaviour in the face of decreased availability of their target species.

Given that diver behaviour can be very influential, the impact of the diver doing the fishing would appear to be very important to observed catch rate trends; and this is in fact what is found in all abalone catch rate standardizations in Tasmania.

This intuition about the influence of divers was so strong that at one point the abalone industry suggested that the current practice of including all divers who have fished in the fishery for more than two years in statistical CPUE standardizations be changed to focus on the CPUE exhibited by the top 30 divers so that those divers whose performance might be less than the more influential divers are excluded. The notion was that these lower performing divers would act to obscure the real catch rates and thereby give an incorrect impression of how the available exploitable biomass was tracking through time. While the intuition that the top divers are likely to have higher absolute catch rates is generally correct it proved to be incorrect to believe they would exhibit different trends in catch rates. In fact, on analysis it was found that the trend exhibited through time was not materially altered by the inclusion or exclusion of the bottom 70% of divers.

The aim of a CPUE standardization is to generate yearly estimates of relative abundance in which more confidence can be held than in the raw CPUE data because the effects of the different factors included have been statistically accounted for (admittedly with assumptions about the constancy of their relative effects through time). This assumption is important when the relationship between effort and catch is considered. If that relationship changes significantly through time for whatever reason, it can be said that the catchability of abalone has changed. The catchability is merely the name given to the concept of the proportion of the available biomass that is taken by one unit of effort under constant conditions. Of course conditions are not constant (different years, divers, areas, and months; other factors such as depth of fishing are not currently available though that may change with the advent of depth loggers on divers) and accounting for this variation is the object of statistical standardization. Typical standardizations have limited data concerning various factors that might influence CPUE, but generally the important factors that influence abalone catch rates are the year and month of fishing, the divers fishing and the areas in which fishing occurred. If other factors for which no data are available have a big impact then the assumption is made implicitly that their effects are random through time. However, as was pointed out in the characterization of the fishery, the fishery changed its character markedly between 1995 and 1997 (see section 7.3.2, p.25; especially **Figure 1**). The relationship between effort and catch changed strikingly and while the divers involved certainly had something to do with that they do not account for the whole change. An unaccounted for change in the catchability across the years 1995 – 1997 remains as a possibility. It may be that there are really two time series of catch rates and not one, which suggests it may be safer to move forward using only the data since 1995. This would certainly warrant further investigation.

Another unexpected finding was that the catch rate trends exhibited by the fishery in the west remains similar in the different areas along the coast, which means that despite rather different catches being taken from different areas the same catch rate is expressed everywhere. Similarly along the east coast while the CPUE trend is different to the west the same trends up and down are expressed in different areas, again despite the catches from those areas being very different. Along a given coast the absolute catch rates do differ by

area but the underlying trends remain the same. For the trends in the available exploitable biomass to vary in this manner on such a scale suggests that the recruitment dynamics, though undoubtedly variable must also be reasonably consistent through time along each coast. Such large scale consistency does not appear to have been noticed or reported before (although it may well have occurred).

The question still remains of whether or not CPUE provides a good indicator of relative abundance of whether there is a high degree of hyperstability affecting the relationship between CPUE and exploitable biomass. Fortunately there are analyses which can be conducted that can give insights into this question.

The range of conditions through which the Tasmanian fisheries have moved is relatively wide with large catches and catch rates followed by low catch rates with similarly low catches (see **Figure 45**, p.158, and **Figure 46**; **Table 15**). Such variable conditions reflect wide differences in fishing mortality rate and stock size which is ideal for producing CPUE data with contrast. While this may have been disturbing to the fishery itself it is very helpful to stock assessments because if there is a close relationship between catch rates and stock size then there are predictions we can make about how a stock should respond to fishing pressure that can be looked for. Thus, if catch rates are informative about stock sizes and their dynamics in response to fishing then the expectation is that increasing the rate of removal of biomass by increasing catches should lead, after a time lag of some years reflecting recruitment and growth in the area, to declines in catch rates and equivalently reducing catches should enable catch rates to increase. This is suggesting that catch rates are a reflection of the balance between the production of biomass into a population by recruitment and growth and its removal by catches; if the removals are less than the inputs then CPUE will rise, and if catches are greater than production then CPUE will fall. This pattern of rise and fall occurs on both the east and the west of Tasmania, where reductions in catches were followed by increases in catch rates and increased catches also appear to have led to decreases in catch rates. If the time lag between these events is consistent through time, this would suggest both that there is a link between catches and subsequent catch rates and therefore that catch rates are reflecting the dynamics of the fished stock (see section 17.6.4, p.168).

In fact, relatively strong negative relationships were found to exist between catches and subsequent catch rates. This means that low catches would lead to increased catch rates in the future and high catches would lead to lower catch rates in the future (see **Figure 56** and **Figure 57**, p.169; **Table 21** and **Table 22**). For the west coast Tasmania, blocks 9 – 12 the best fitting time lag proved to be seven years, while for each coast blocks 13 – 31 this was only five years. There was a wider range of time lags indicating significant relationships on the west coast than on the east although the optimum fit was relatively well marked in both cases. The east coast exhibited two cycles that followed the time-lag model while the west coast only exhibited a single cycle.

The major results from the examination of CPUE as a performance measure were that the same catch rate pattern was found across relatively large geographical areas, even where the catches were rather different. This was unexpected and was suggestive of recruitment dynamics being more stable than previously imagined. Secondly, the range of conditions under which the Tasmanian fishery has operated has been relatively wide, which has made things difficult for the fishery but has been very good for introducing contrast into the fisheries data. For this reason formal stock assessment models can fit the CPUE data quite

well. Finally, there are reasonably clear relationships between catch levels and subsequent catch rates that suggest that CPUE is a useful reflection of the relative abundance of abalone stocks, at least around Tasmania.

These outcomes indicate that the use of abalone fishery dependent catch rates can certainly be informative about the relative status of the stock and that catch rates could indeed be used as an empirical performance measure. The fact that there are time-lags, however, means that there could be real delays in management actions (controlling the catch levels) influencing the performance measure being used to recommend those management actions. This is not surprising as changing the catches are likely to be influential on the amount of spawning biomass available, which will influence the recruitment levels. But for that to influence catch rates the new juvenile abalone need to grow through the LML and become available to the fishery. It is also not surprising that there were a range of years over which a significant influence of catches on catch rates could be detected. The growth of abalone is not deterministic and some animals would take less and other more than an average number of years to growth up to and through the LML.

7.4.2.4 *Catch Length Structure*

Since 2008 data logging measuring boards have been used in processing sheds to measure relatively large samples of landed catches. With sample sizes in the 1000's this has provided much better estimates of the size distribution of catches than were previously obtainable (see section 17.8, p.172; **Figure 58**, **Figure 59**, and **Figure 63**). This means there are only five years of high quality data.

In the south west (subblocks 9A – 13B) the five years of data only exhibit relatively minor changes in the observed size distribution of catches through time but as the relative change in standardized catch rates over that time is also not large (see **Figure 60**, p.173 and **Figure 61**) this is not a useful or informative result from the point of view of determining the value of length frequency data as a performance indicator. Fortunately, from the point of view of this study, on the east coast, in the Actaeon area there has been a relatively large decline in catch rates from 2008 to 2012 (see **Figure 62**). Remarkably, even though the stock on the whole of the east coast is clearly in a depleted state there has been only very minor changes in the length frequencies taken in the catches (see **Figure 63**).

While the length frequency information of the catch can have value in a formal stock assessment model it is clear that as an empirical performance measure acting on its own the size distribution of the catch is not sensitive to large changes in the stock status and so would not be expected to contribute usefully to being included in a decision framework reliant on only empirical information directly from the fishery.

7.4.3 Model Based Performance Measures

7.4.3.1 *A Difficulty with Models*

Mathematical models are of necessity an abstraction of whatever is being modelled. This means that, hopefully, important influences are included and those that have insignificant effects on the population dynamics can be omitted. Unfortunately, with abalone stocks, even if some of the spatial heterogeneity observed is simply due to sampling errors (Heli-doniotis and Haddon, 2013) there remain large scale variations in productivity in relatively close geographical proximity which clearly reflect differences between populations. Omitting such obvious spatial differences from stock assessment models suggests that the average behaviour across the geographical extent of the stock being assessed is sufficient

for the generation of adequate and safe management advice. This is an important assumption and it should be emphasized that stock assessment models need not be exact reflections of the dynamics of the populations they profess to represent. What actually matters is whether the management advice that comes from the models doesn't mislead the managers into damaging the stock or missing out on significant catches.

In Australia, formal stock assessment models have been used in New South Wales and in Victoria. Victoria has had issues with disease and mortality events, which is a challenge for any model. NSW appears to have been sustainably over-fished for a long time and this was not detected by the assessment model, possibly because the contrast in the data going into the model was not very great. It was certainly the case that catches have varied over a wide range but the stock had not been depleted and then recovered as in Tasmania, and that type of scenario is optimal for introducing contrast into catch rate and other data streams. It would be unfortunate if the potential value of formal stock assessment models was discounted because the available data that had been put into them had not allowed those that have been used in Australia to be useful.

In Tasmania, formal surplus production models and size-based integrated assessments have been implemented in the south-east and south-west (for example, Haddon, 2009, and Haddon, 2011). However, because of doubts about the representation of growth in the size-based model, and the fact that there was no management context in which to use its outputs, these have only ever been used for strategic work rather than for producing management advice. For example, size selective fishing became an issue in the early to mid-2000s and an earlier implementation of Haddon (2009) was used to demonstrate that such fishing behaviour was self-defeating and must lead to a reduction in the TAC if the full size range of the potential catch was not taken.

Nevertheless, such models can still give an indication of the potential production that can be expected from different stocks. Unfortunately, few models can accommodate random, rare events, such as a mortality event that occurred at least in the Actaeon area in March/April 2010. Nevertheless, when data has the contrast expressed by the Tasmanian CPUE data even simple models can lead to some appreciation of how the stocks can be expected to respond to changes in the fishery. One large assumption in all such work is that the dynamics of the stocks involved remain similar to how they have operated in the past. With the advent of climate change induced alterations to sea water temperatures (and other environmental factors such as pH) this assumption may become compromised, however, that is a complication that will require further observations and work.

7.4.3.2 *Surplus Production Models*

An analysis of Blocks 9 – 12 from the south west of Tasmania was conducted and with data up until 2011 the analysis here goes slightly further than has been investigated in Haddon (2011), although many more details and greater consideration given to uncertainty in the latter. Because there is no attention paid to size or gender or any such details the effect of a change in LML, as occurred in 1990 (132mm – 140mm) can be approximately by introducing a change in the catchability (which is used to relate catch rates to exploitable biomass; see section 18.2, p.178).

In the case of blocks 9 – 12 the estimate of MSY was only about 850 t, and catches were greater than that between 2000 – 2008, and have only been about 842 t between 2009 – 2012. Of course, it would be better to conduct the projections either from bootstrap repli-

cates or from MCMC parameter sets under different proposed catch levels, but if rebuilding is wanted, it suggests that catches are currently too high so CPUE might be expected to continue to decline albeit relatively slowly.

Because of the strong relationship between catch rates and the stock status, at least in Tasmania, these models are candidates for further use in Tasmania.

7.4.3.3 *Size-Based Integrated Assessments*

The data requirements of size-based integrated assessments are much greater than that of surplus production models, however, the potential outputs can be more varied (see section 18.3, p.186). The increased range of potential outputs opens opportunities for using a much wider range of performance measures derived from the model outputs, such as the state of spawning biomass depletion, the relative reproductive output, the annual fishing mortality rate, and others (see **Table 25** and **Figure 66**). Of greater importance, as with the surplus production modelling, is the capacity to project the stock dynamics forward under different management arrangements. Because fishing selectivity is included explicitly in the mathematical description of the dynamics, in the case of size-based models both management options of changes in TAC and changes in the LML can be projected forward to determine the likely outcome (see section 18.3.5, p.193; **Figure 71**). These model projections are simpler than with management strategy evaluation as they involve simply imposing the potential management option and projecting the outcome forward without feedback into the dynamics (which requires the generation of simulated fishery data, the simulated assessment of that data, the use of a simulated HCR and then implementing any suggested changes into the management). Despite being relatively simple such “risk assessment” projections, which include variation reflective of the dynamics of the system being projected, do enable the relative risk of the alternatives being considered achieving the aims of the management (see **Figure 68** and **Figure 69**).

The aims or objectives of such risk assessments also need not be complex and, for example, can include such things as: the stock biomass will be larger than currently in five years’ time with a probability greater than 70%. An important component of such risk assessment projections is the inclusion of probability levels in the objectives to be achieved. This is what provides a criterion for determining which management option or options will be expected to meet the stated aims for the fishery. A 50% chance (see **Figure 68**) may appear to provide for a stable fishery but the variability around any outputs implies that while 50% of runs may be above the average line there are also 50% that were below the objective aspired to for the fishery. The ability to conduct such risk assessments projections that include the implications of changing the LML is a clear advantage over using surplus production models. This advantage is important because, at least in Tasmania the two classes of models produce remarkably similar outcomes with a size based model also suggested that the MSY for blocks 9 – 12 in the south-west is about 850 t. This model could thus be used to predict the likely effect of changing the LML in the west to 145 mm instead of 140 mm, a management change that has recently been suggested. It would appear that the CPUE data is dominating the outputs of both types of model.

The data and information requirements of size-based models are significantly greater than for the simpler surplus production models. Surplus production models are only appropriate when there is contrast in the catch and effort data, meaning there is data available from a wide range of stock sizes and of different catch levels. In addition, it is necessary that there exists a relatively simple relationship between catch rates and the amounts of ex-

exploitable biomass available to the fishery. These requirements are also necessary for the application of size-based models to be appropriate and these mainly revolve around the available data being representative of the assessed stocks. However, there are other requirements that apply only to size-based models. The growth description that is used needs to be representative, which can be difficult to achieve where the growth patterns appear to be highly variable.

The growth models used relate to descriptions of the expected growth increments for different size classes. These are best determined using tagging data although a question remains whether the tagging process influences the growth expressed or perceived. It was clear on the west coast of Tasmania that the tagging data provided a good estimate of growth where the samples were taken but for some reason the growth it implied was insufficient to explain the observed dynamics from the fishery. A solution was found, that used the commercial catch at length data to correct some of the growth parameters when fitting the model, and this permitted the size-based model to be fitted to data successfully. However, uncertainties remain about the representativeness of growth when estimated using tagging data, and these need to be clarified before such size-based models can be used with confidence. It is possible that the tagging data was obtained in relatively sheltered locations which did not permit the abalone to fully express their potential for maximum growth and so were only partially representative of most of the areas that are harvested. Alternatively, it could be the process of tagging abalone itself that leads to the tagged animals having a disadvantage in growth and ending up only expressing a proportion of their potential. This is still under investigation; the difficulties of conducting a tagging experiment on abalone on the open coast should not be under-estimated.

Of similar importance is the use of the λ parameter in the relationship between catch rates and exploitable biomass, as in equations (10) and (38). By setting λ to 1.0 this implies that the relationship is linear, which in turn implies that catch rates are directly related to exploitable biomass and that hyper-stability of catch rates is not an issue. While the empirical exploration of CPUE in Tasmania (see section 17.4, p.148) indicates a strong relationship between the two there it should also be examined elsewhere to see if this assumption holds.

All of these assumptions and options remain open to further exploration. As with surplus production models, size-based models remain open to further development and use in Tasmania. Their failure elsewhere should not be reason to dismiss their possibilities in Tasmania where at least CPUE data is informative.

7.5 Design a MSE Framework suitable for Abalone

7.5.1 Introduction

The provision of a ‘national MSE framework’ was intended to imply the production of a general abalone MSE simulation framework that could be adapted to suit any of the jurisdictions within Australia. It was explicitly not meant to imply that all jurisdictions would be involved in the technical design of the simulation framework. A single MSE framework common to all jurisdictions is simply not possible given the diversity of data structures and performance measures available in each jurisdiction; this is why the strategy was adopted by developing a flexible and adaptable framework that could be customized relatively easily to match each jurisdictions idiosyncrasies.

The production of a simulation framework for abalone populations to conduct management strategy evaluation of management options in the abalone fishery was a more difficult proposition than first imagined. For it to be based where possible on real abalone population biology and properties, rather than mostly on plausible guess work, it was first necessary to extract and analyze data from as many separate populations as had been sampled from around Tasmania, which is where most data was made available. Some of this work had been done already, especially for size at maturity work (Tarbath et al, 2001) but the majority of populations and biological properties had not been examined in detail. Fortunately the structure of the database in which this data had been held was improved in about 2004/5 through an initiative by Dr Craig Mundy and this enabled the extraction of large amounts of data at once. The use of the statistical software R (R Core Team, 2013) also enabled the multiple analyses required to be conducted in a relatively straightforward fashion that would have been difficult previously. For example, over three hundred maturity ogives were fitted, numerous growth curves, weight at length, and other analyses were required.

The growth description is so influential on the dynamics that special attention was paid to that. A review growth curves previously used with abalone, which was conducted during the development of the proposal that led to this work, quickly led to the conclusion that current models only provided inadequate descriptions of abalone growth. They may have provided reasonable descriptions of part of the growth trajectory of abalone but a growth equation was required that was able to capture growth dynamics across the full range of sizes experienced by abalone during their life-cycle. This was necessary because growth takes time an important aspect of the dynamics in any population dynamics are the time lags that exist between different life stages. If the time-lags are mis-represented (in the worst case, omitted), then the resulting dynamics would become biased and lead to incorrect expectations of what is possible. This work led to the development of a new Inverse Logistic growth curve (Haddon et al, 2008), on which work has continued (Helidoniotis et al, 2011, Helidoniotis and Haddon, 2012, 2013).

Once such information was available this provided a much better appreciation of the spatial variation that existed in Tasmania which, in turn, suggested how to structure the simulation framework.

7.5.2 The Model Structure

Not surprisingly the operating model used to describe the dynamics of a simulated abalone zone is made up of multiple populations, and its related fishery; the operating model has numerous components and appears relatively complex. Nevertheless, a simulated abalone zone has an hierarchical structure where a zone is made up of a number of spatial assessment units (SAUs), the statistical blocks of Tasmania, and these can contain a number of abalone populations of varying size, productivity, and other properties (see section 19, p.199; **Figure 73**).

The fundamental unit within the operating model is therefore the population and this forms the basic building block within the simulation framework. To define a zone it is therefore necessary to define the number of populations across the complete zone, and then the number of SAUs these populations are to be grouped into and finally how they were to be grouped, that is, which population was to be a part of what SAU. For ease of calculation and consequent summary, each population, in the software, was designed to carry all the information required to define and characterize it (**Table 28**).

The spatial assessment unit structure (statistical blocks in Tasmania) was imposed on top of the population structure and the zone's properties, as well as its fishery, could be examined at a population level, at an SAU level, or at the whole zone level. The simulated data from the framework that could be used in any chosen assessment included the catch, the effort, the catch rate, and the spatial distribution of those statistics, and the size distribution of the catch (with pre-specified levels of uncertainty being included in such simulated data). These are the classical performance measures available in all Australian abalone fisheries; the size distribution of the catch is generally the least well known, although exceptions do exist where measuring boards are used as data loggers.

When comparing the performance of different management strategies it is also necessary to consider the state of the underlying stock. The data used to examine the state of the resource include the fishery data above but also the exploitable and spawning biomass (spawning and mature biomass are terms used interchangeably), the related harvest rates, the size distribution of the populations (rather than that of the catch), the recruitment distributions, and the relative distribution of such things spatially. When not dealing with fisheries data there would be no uncertainty added to the available data.

The conditioning of the operating model has already been described (see sections 7.3.4 and 19.4, p.209).

One remaining important source of uncertainty relates to the recruitment dynamics within abalone populations. Currently, for each population, recruitment has been implemented as a classical spawning stock and subsequent recruitment relationship is used with a relatively low ($h = 0.5$, with variation) steepness, which means that reductions in spawning biomass can be expected to reduce the average recruitment levels in a noticeable manner. But on top of that, has been added an occasional doubling of settlement success across a whole zone (only one change in 25 – 30 of it occurring). Similarly, for each population, there is an occasional failure to recruit so that settlement is only a tenth of expected levels (one change in 25 – 35). These probabilities are very approximate estimates taken from the Tasmanian fishery where an occasional cohort is not found in an area (Helidoniotis and Haddon, 2012), which was taken to imply occasional local recruitment failure or possibly mortality events. In addition, CPUE trends have been found to be similar across large geographical areas and the recovery on the east coast occurring remarkably rapidly, which was taken to imply that occasionally there are large scale successful recruitment events that can influence dynamics. These additions have been included because without them the stocks sometime have difficulty in recovering as quickly as they have been observed to do. In addition, the simulated dynamics are less variable than have been observed.

While these extras added to the recruitment dynamics, which can easily be turned off in the MSE simulation framework, have some basis in evidence it remains circumstantial evidence. The details surrounding recruitment are clearly in need of further clarification, which could take the form of both observations and further simulation studies to see the influence of plausible combinations of events.

One great value of modelling, which has not received any attention yet, is the fact that a model usually synthesized what is known about a species and its biology and population dynamics. If something about the dynamics of the model fails to reflect wild observations well this indicates that something is missing or incorrect. Such models automatically act

as a way of identifying high priority research areas, and currently growth and recruitment remain high priorities.

7.5.3 Productivity of a Simulated Zone

By setting the random variation that was added to recruitment and to how catches were distributed about the zone, and to the simulated observations on CPUE used in the HCR, it was possible to approximate the deterministic behaviour of the simulated populations, statistical blocks and finally the zone. This was useful because it enabled things like the optimum sustainable productivity to be estimated for the simulated zone (or for any of the smaller scale areas), which places a limit on the potential yield that would assist in explaining the response of the simulated stock to the level of catch removed (see section 19.6, p.218).

In addition, to assisting with understanding the outputs from the simulations the shape of the yield curve also lends insight into the behaviour of abalone fisheries in general (**Figure 5**).

Because part of the spawning biomass is protected by the LML, while the exploitable biomass is everything larger than the LML, the exploitable biomass will always be less than the spawning biomass.

The theoretical productivity from the simulated zone as a whole, 632.8 t, is smaller than the theoretical combined MSY from each of the component 70 populations (which in the example used in the simulations here was 662.163 t). This is merely a reflection that not all populations have equivalent dynamics and that the distribution of catches is made via the relative distribution of exploitable biomass across the statistical blocks of which the populations are members, so the optimum catch from each population is not necessarily taken. Importantly, the yield produced by the zone is relatively flat across a wide range of harvest rates and consequently of catches and depletion levels.

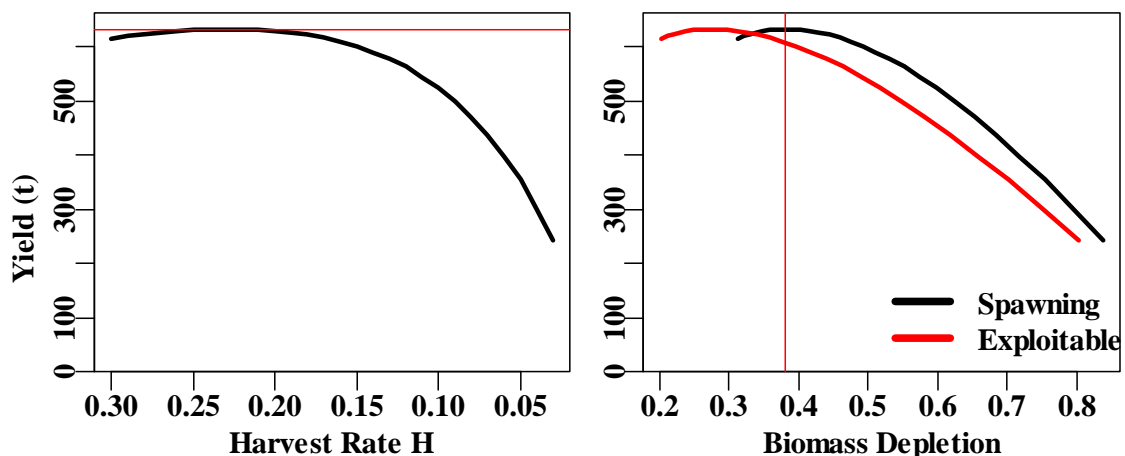


Figure 5. The theoretical productivity of a simulated zone of 70 populations with an overall BML of 138mm. The biomass depletion relative to the unfished state and the straight red lines depict the MSY at 632.8 t. Copy of **Figure 83**.

The flatness of the yield curve means that the harvest rate could be set too high for a long time and even though the resulting catch levels would not be sustainable, and would certainly require more effort than was actually needed and so be less profitable than it could

be, it might take a long time for significant depletion from such unsustainable catches to occur. This is both good, because short term excesses need not do too much damage, but also bad, because it would be difficult to know that certain harvest rates were not sustainable even though they might appear to be over a long period. This is a potential risk for abalone fisheries that is dangerous because it would be difficult to detect when it was occurring.

7.6 Legal Minimum Length and Total Allowable Catch

7.6.1 The Context

An important management tool is the Legal Minimum Length (LML) used to give some protection to immature and young animals and to act as insurance and allow at least some spawning biomass to be safe from fishing mortality. While this approach is used extensively in many fisheries, especially with invertebrate fisheries there doesn't appear to be any standard method for selecting the most appropriate size at which to set the LML for a particular fishery. In many cases market preferences rather than biological considerations have led to size limits being set (the *Katelesia* clam fishery and purple wrasse fishery in Tasmania are two examples). In Tasmania, possibly as a post-hoc justification for the relatively arbitrary LML of 5 inches selected early in the fisheries history, the idea of setting the LML at a size representing two years' growth following maturity was introduced (perhaps following a similar idea used in the scallop fishery at a similar time). This led to many samples being taken in an attempt to estimate the size at maturity of different populations around the state, and eventually led to changes in the LML in various different places. The first major change, from 127mm to 132mm occurred in 1987 and was mostly driven by the depleted state of the resource being recognized by all stakeholders. More recently changes and proposed changes have been led by the size at maturity sampling.

However, whenever these are suggestions to modify the LML there is always debate over the potential gains and losses. The MSE simulation framework was used to explore the implications of making such changes (see section 20, p.221).

7.6.2 Implications for Yield and Spawning Biomass Protected

As the LML is increased for any population or stock then the expectation would be that the proportion of the spawning biomass protected would increase, but exactly by how much it would increase is unclear. At the same time it is unclear whether the amount of yield that it would be possible to take sustainably from the stock would remain the same or increase or decrease (see sections 20.3.2, p.223, 20.3.5, 20.4.3 and 20.4.4).

The question of the proportion of spawning biomass protected is not as simple as it might seem. The method used considers the unfished biomass and size distribution and then it is simple to sum up the numbers at size above the LML and convert that to weight and compare to the total. However, when a stock is fished, especially abalone, in which a reduction in spawning biomass are assumed to lead to reductions in subsequent recruitment, while it is expected that the number of animals above the LML will decline what may not be expected but will happen is that the number of animals below the LML will also decline. In this way the proportion of the original unfished biomass protected may decline. Exactly what happens to the proportion of available spawning biomass as fishing continues is more complex and how best to represent such complex changes is not yet clear. This is, perhaps, something that could be explored further in later work.

The Biological Minimum Length (BML) is defined in this report as the average length across a zone at which each population has had two years' growth following its size at 50% maturity. In terms of the unfished biomass, not surprisingly the closer the LML is to the BML the more the spawning biomass is protected (see section 20.4.3, p.228; **Figure 88** and **Figure 89**). As it turns out the two year rule appears to provide about 20 – 24% protection to the spawning biomass in an unfished population. It also provides greater resilience to the fished stock as its maximum productivity occurs at a higher level of spawning biomass. When fishing a below the maximum productivity the extra protection afforded by a larger LML is reduced because a large proportion of the spawning biomass remains above the LML, as one would expect it is only when the biomass above the LML starts to become more depleted that the effectiveness of the LML begins to become fully expressed (see **Table 33**, p.231, **Table 38**, p.249, and **Table 41**, p.267).

Ideally the LML should be related to the size at maturity of each population but a compromise is less risky than setting it too low. Using the two year rule appears to be a reasonable guideline when a LML is being set across large geographical regions. Setting it higher or lower than the level from the two year rule, the BML, are both poor options for different reasons. Setting the LML too high and yield and potential quality of product are sacrificed, setting it too low and it is possible to deplete the populations to a low level. If a recruitment failure then occurs, especially if it occurs more than once in a row, and yet fishing continues there may be such an absence of new entrants growing above the LML that the fishery collapses and the lack of spawning biomass locally leads to an extended period of fishery and stock collapse. If the habitat changes during such a low abalone population period, the population may never come back in anything like its previous abundance.

In terms of potential yield it appears that any differences in maximum yield possible from a given stock at different LML are only minor (**Figure 6**; and see section 20.4.4, p.229; **Figure 90**), at least within a zone having a particular BML.

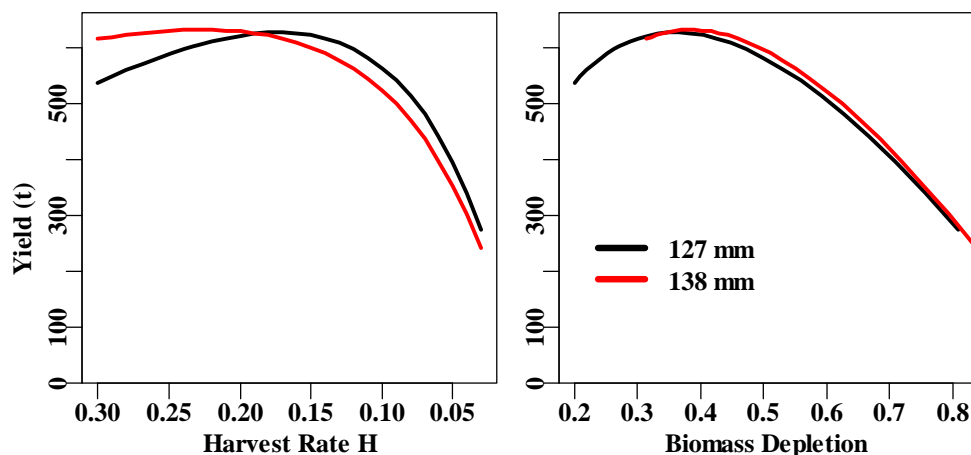


Figure 6. A comparison of the effect of two alternative LMLs (127 and 138 mm) on the productivity of a single stock with an average BML (the size at two years' growth past the size at maturity) of 138 mm.

Increasing the LML until it is at least close to the BML appears to have many advantages, however, there remains a trade-off, such that while the maximum yield remains roughly

the same in order to achieve that production the harvest rate will need to be higher in order to gain the same level of catch in tonnes and this will take more effort for possibly a lower catch rate (**Figure 6** and **Table 38**). The trade-off is thus greater protection and resilience against unpredictable events (and inadvertent over-fishing) against somewhat lower efficiency and lower catch rates. The decrease in CPUE is likely to be mitigated by the fact that any abalone taken will be larger and therefore heavier. The difference in numbers required to capture the same weight near the maximum productivity can be between a 10 – 12% increase if the catch is taken at 127mm rather than 138mm (see section 20.4.5, p.230; **Table 33**). While the catch rate does decline somewhat, the size distribution of the catch also, obviously, changes, so the full economic effects would also be determined by the market preference for particular sizes.

Importantly, this suggests that it would be tempting to lower LML when catch rates become relatively low so as to increase CPUE again. However, if the reduction in CPUE is due to depletion of the stock or a lack of successful recruitment then lowering the LML would reduce any resilience to whatever has been perturbing the stock (be it over-fishing, a mortality event, or a disease event) and would increase the risk of not only the fishery being in trouble but the underlying stock failing also.

7.7 MSE Testing of Alternative Harvest Control Rules

7.7.1 The Need for Objectives

The previous sections detail the design and production of a simulation framework for testing management strategies in abalone fisheries and it has been used to examine the trade-offs that occur when managing abalone fisheries using TACs and LMLs. Management Strategy Evaluation (MSE) frameworks are characterized by having feedback mechanisms simulating the fisheries management arrangements in place that take the outcome of management each year and have that influence subsequent management. To take full advantage of that requires that we work with and define full management strategies that include the data used, the assessments used to produce the performance measures that represent the state of the fishery, and the harvest control rules that translate the changes in performance measures into management advice.

For testing the efficacy of the currently used management (or harvest) strategies we have focussed on catch and CPUE data as that currently appears to be the dominant sources of information about each fishery that influence the management. The performance measures have involved the absolute catch rates relative to previous catch rates and also, more specifically, recent trends in catch rates. However, these considerations have been mostly qualitative and the production of management advice has involved attempts to obtain consensus over what would be a sensible expectation of catch or yield from different areas.

In 2012, new management plans and formal harvest strategies were introduced into South Australian abalone fisheries. These saw the advent of formal harvest control rules that had the form of a multi-criterion decision analysis that could include catch rates, catches, survey information relating to density on the ground of different size categories, and information from size distribution data from the commercial catch (Chick and Mayfield, 2012; Stobart et al, 2012). However, until this new harvest control rule (HCR) is better known and its operation proceeds effectively, the relatively informal assessment that has been previously used in South Australia is also being run in parallel. This latter is fortunate be-

cause a number of issues did arise in the first run of the new HCR and changes are already being contemplated. The same kind of HCR is being proposed for Tasmania but the intent there is to test it where possible first before using it for management.

To do this required that the relatively informal use of the catch rate performance measures be translated into something more formal and then combined with a multi-criterion decision analysis (MCDA) framework. The MCDA effectively ranks the degree of change expressed by the performance measures used within it (the criteria) and produces a scaling factor to be applied to the TAC to determine whether it should increase, stay the same, or decrease. Only single criteria were included in the arrangements considered with the MSE simulation framework. This was for clarity of outcome and ease of interpretation. Further complexity will be introduced in future work. These combined arrangements were referred to as separate harvest control rules (HCR).

7.7.2 The Harvest Control Rules

Two families of HCR were developed for testing (see section 21.2, p.240). The first used the gradient of a given number of years backwards from the present. In simple terms, if the gradient of recent CPUE (scaled to a mean of one) increased or decreased more than a certain amount then the TAC increased or decreased in proportion to the gradient expressed. The second defined a target CPUE towards which the fishery was managed by changing the TAC up or down appropriately, so if the current CPUE was below the target a decline in TAC was recommended, while if above the target an increase in TAC was recommended. The extent of the change in TAC depended in how far above or below the target the current CPUE was found to be. In both of these cases, random variation in the expressed CPUE from the fishery was clearly going to have an effect.

The first HCR relating to CPUE gradients appears to have some things in common with current practice in that no specific target is made explicit. It simply reflects a feeling that declines in catch rates are sometimes a bad things but that rises in catch rates is a good thing. Falls in catch rate are not necessarily to be avoided if the stock starts off at what is agreed to be a high level. What this means is that the HCR, as it is expressed in the current work, does not reflect that the absolute level of CPUE from which a CPUE gradient is calculated is also taken into account. This has been expressed on the west coast of Tasmania where following zonation higher catches were taken from the west but the average catch rates in 2000 were about 156 kg/hr and they took 6 years to become about 125 kg/hr in 2006. Even though there was a steady decline in catch rates during that period, it was not considered a problem because the base CPUE was so high relative to elsewhere in the fishery. However, the decline has continued to about 110 kg/hr in 2012 (**Table 1**).

7.7.3 The CPUE Gradient HCR

The CPUE gradient HCR, as implemented in the tests was surprisingly ineffective (see section 21.4.1; **Figure 103** to **Figure 115**). The starting conditions under which it was tested were reasonably broad covering three LML, three initial depletion levels, and three initial TACs, but, very broadly, its behaviour only reflected initial conditions. At best the CPUE gradient HCR was a status quo management tool that could lead to oscillations in spawning biomass and catch rates, but also to much greater oscillations in catches taken through time. It did not demonstrate any ability to recover a depleted stock nor did it exhibit the capacity to manage a rational fish down of an un-depleted stock. Worst of all, in many instances, especially those where the stock is under the stress or too great a deple-

tion or too high a catch level (overfished or overfishing), then slow depletion appears to occur, sometimes so slow that it would be hard to detect (**Figure 106** and **Figure 107**).

Table 1. Summary statistics from the logbook database for Tasmanian western blocks 9 – 12. Catch rates are geometric mean catch rates rather than arithmetic.

Year	Catch	CPUE	Records	Effort
1985	1018.884	73.565	2116	12408
1986	742.347	78.223	1477	8540
1987	868.023	78.311	1721	10076
1988	715.104	79.805	1345	8018
1989	585.651	82.151	1060	6164
1990	532.214	82.990	996	5741
1991	566.507	88.245	977	5859
1992	611.126	91.477	1017	6349
1993	548.256	100.152	853	5050
1994	499.406	106.578	773	4316
1995	478.919	120.656	668	3662
1996	427.787	131.610	546	3052
1997	657.496	143.157	804	4373
1998	480.660	148.531	591	3038
1999	645.049	156.693	806	3842
2000	973.291	156.776	1264	5747
2001	968.369	143.440	1375	6212
2002	925.547	140.795	1316	6043
2003	972.389	138.733	1455	6463
2004	962.235	129.350	1557	6870
2005	940.518	124.038	1514	7087
2006	953.586	125.450	1577	7108
2007	1030.043	122.266	1790	7938
2008	984.599	111.036	1789	8315
2009	835.576	121.300	1463	6480
2010	839.916	117.536	1492	6679
2011	833.496	117.990	1410	6678
2012	858.176	109.975	1580	7246

The scenarios based on initial TACs of 800 t, exhibit the greatest variation and the outcomes with the largest oscillations (**Figure 106** and **Figure 107**). By considering only those scenarios it becomes clear that in all cases where the initial TAC was 800 t the spawning biomass gradually declines through the 40 years of application of the CPUE gradient HCR. The greatest rates of decline of spawning biomass occur with the higher levels of initial depletion. The effect of the TAC limit on changes to the spawning biomass is to exacerbate any declines that occur.

Although catches oscillate, in all scenarios with initial depletion levels of 50% and 40% the catches oscillate around some long term average that is approximately 600 t, this also occurs for the 30% initial depletion and the 127mm LML. However, the 30% initial depletion at the LML of 132mm and 138mm leads to stronger oscillations associated with declining catches. The oscillations in catch are especially large with the 132mm LML in the no TAC limit version. While this appears to suggest that the TAC limit is a good option, the positive benefits are only in terms of catches. The trade-offs would be in terms of very

low catch rates and a greater risk of reducing the amount of spawning biomass (**Figure 106**, p.260, and **Figure 107**).

Even though the simulated base zone had a BML of 138mm the optimum LML appeared to be 127mm, which led to the smallest oscillations in spawning biomass, CPUE, and catches across the different combinations of initial depletion and initial TAC (**Figure 103**, **Figure 104**, and **Figure 105**). However, in the scenarios depletion was fixed without note being made of CPUE. In practice, one would expect the management to respond in the LML situation if the CPUE declined below previously acceptable levels. When a simulated zone, being fished at different LML, was depleted to the same CPUE level, in all cases, the larger the LML, the greater the spawning biomass. What this means is that there are risks if an abalone fishery is managed by simply increasing a LML while not managing TACs as well. Increasing a LML does not mean that the zone is secure even in the face of excessive over-fishing. The advantage of increasing the LML is to increase the chance that the same catch can continue to be taken.

The current oscillations in catch rate on the east coast of Tasmania are not a positive sign. It suggests that the eastern zone has been fished too hard and that allowing the TAC to increase again to previously high levels once the CPUE reached previously high levels, merely led to a repeat of depletion and a continuation of the oscillations, which are beneficial to nobody.

If the CPUE gradient started off relatively flat then this HCR operated as a status quo strategy. Unfortunately, it didn't matter whether the stock started at a high or a low level, the HCR operated to keep it where it started.

In its current form the HCR cannot be recommended, however, the outcomes appear more positive if the response to the gradient is made to be asymmetric (see section 21.4.3, p.266; and **Figure 116**), thus, while catches do increase if the CPUE gradient is positive, they do so at only a fraction of the decrease when the gradient is negative. When this modification is made then the HCR appears capable of recovering a depleted stock with a rapidity that depends on the degree of asymmetry. However, the HCR in this modified form still fails to manage a fish down of an under-fished zone. When examining multiple performance measures in a single multi-criterion decision framework then this modified form of the CPUE gradient HCR should be considered as a component for inclusion.

7.7.4 Target CPUE HCR

The essence of this HCR is that a target is defined towards which a fishery can be managed. The obvious advantage of this is that the performance of management can be determined. Before exploring the dynamics associated with this HCR the basic diagnostics available were characterized. By running a single zone in the effective absence of variation almost deterministic behaviour can be produced and this can be used to search for initial conditions that give rise to the expected potential yields and potential catch rates at those yields (see **Table 41** and **Figure 117** in section 21.4.4, p.267). Because these are simulated zones this actual value is not really important as it is relative performance relative to the target that relates to efficacy of the HCR, but the empirical MSY, given the same zone, appears to be about 635 t for each of the LML. The catch rates at the MSY appear to be similar with slightly higher catch rates at the larger LML, and, not surprisingly, the spawning biomass also increases with LML (**Table 41**).

Within each scenario the pattern of response of the catch rates and the spawning biomass were very similar, which merely illustrates that the exploitable biomass tends to follow the same trajectory as the spawning biomass (although the ratio of exploitable to spawning biomass varies with LML). The pattern of response of catches to the initial TAC (iTAC) and initial depletion level was similar under each of the three LML although the depth of any declines and the heights of any rises increased with LML (**Figure 118**, p.269, **Figure 119**, and **Figure 120**). As expected, given the empirical MSY values the initial TAC of 450 t values tend to lead to relatively high spawning biomass levels and related catch rates at the end of the initiation period, which in turn lead to increases in catches; although exceptions occurred when an iTAC of 450 t was combined with the two larger LML and lowest initial depletion levels. The increased catches are initially associated with increases in catch rates and spawning biomass which then fall after a time lag by which time catches have once more increased beyond surplus production. Surprisingly, with the initial TAC of 450 t, the 50% initial depletion level led to the most variable response in biomass, catch rates and catches across all three LML. This again reflected time lags in the response of catch rates to changes in the catches, which derive from the time it takes for new recruits to grow from the spawning biomass the availability of which depends eventually upon the catch levels.

The most stable trajectories are exhibited with the initial TAC of 600 t. With the LML of 127mm the 40% initial depletion level remains generally flat, with a final ten year mean of 43.4% at an average catch of 574 t. Events were slightly more variable with the LML of 132mm but that still exhibited a final 10-year mean of 44.7% depletion and average catch of 569 t. With the 138mm LML a similarly stable trajectory was only produced by an initial depletion of 50% which ended with a final ten year mean of 48.8% and an average catch of 566 t (**Figure 118**, **Figure 119**, and **Figure 120**).

This HCR certainly appears capable of recovering a depleted stock and of managing an under-fished stock. In all instances it appeared capable of achieving the CPUE target even in circumstances that means it took nearly the full 40 years to do so. Especially with the smaller two LML the target CPUE tended to be over-shot, which again indicates that time-lags are influential on outcomes when using CPUE to manage subsequent TACs.

7.7.4.1 *The TAC Lower Limit Option*

The effect of the lower limit on the TAC is primarily exhibited by the 800 t initial TAC scenarios and any effects appear to be exacerbated by increases in the LML (**Table 42**, p.271). This has consequent effects on the spawning biomass and on catch rates. When the first year of hitting the limit TAC is considered (**Figure 121**, **Table 42**) then the predominance of effects in the scenarios involving the initial TAC of 800 t is clear, although hitting the TAC limit also occurs with an initial TAC of 600 t with the larger two LML. In general the TAC limit was only influential in scenarios under which the simulated zone was stressed, either through over-fishing (catching too much) or being over-fished (being depleted to low levels). The duration of being at the TAC limit also varied across scenarios (**Figure 122**). The larger the LML the longer, in general, the simulated zone remained at the TAC limit once it hit it. Meeting the limit occurred mostly due to high levels of depletion but also occurred occasionally as a result of highly variable outcomes. If the TAC limit were set at some absolute level, such as 400 t, the level implied by 50% of the initial TAC in the 800 t scenarios, then there would have been more instances of its occurrence in the other scenario combinations.

Overall the effect of the TAC Limit was to delay recovery if it occurred and to keep CPUE down. It did avoid some variation in catches, but led to greater variation in catch rates and in spawning biomass. The rapid oscillations seen in the CPUE gradient HCR were not seen.

7.7.4.2 *The Trade-Off between CPUE and Catch*

The target CPUE HCR appears to be of value in producing management advice although catches set at levels either above or below the MSY can lead to more variable outcomes in spawning biomass, CPUE, and catches. If a TAC limit is used then great care is needed to set the associated TAC limit appropriately. The potential trade-off between total catch and CPUE within a set of years is a potentially important question but, unfortunately this only tends to become an issue when a stock is in a depleted state and the fishery is reduced. Keeping catches elevated is a riskier option, as has been discovered in New South Wales (see **Figure 124**). If the limit TAC is set too high and a stock becomes depleted the limit (catches) could prevent a stock recovering. It is possible in the simulation framework to generate an essentially stable outcome with ongoing depressed catch rates (**Figure 125**). This state of low level stable catch rates is reminiscent of the situation that was being expressed in NSW, Australia until TACs were cut to relatively low levels following which there are signs of recovery and CPUE levels unseen for decades. Once again, with empirical performance measures it is not possible to know this response would happen without trying it. Each fishery will need to build up experience and knowledge of its own dynamics.

7.7.4.3 *Years to Achieve the CPUE Target*

Within any particular initial TAC and initial depletion level, as the LML increased the number of years it takes to reach the target increases. On top of this, if there is a TAC limit this also increases the number of years to reach the target (**Table 44**, p277, **Figure 126**).

Only in the case of the initial TAC of 800t, an initial depletion of 30%, and a LML of 138mm were there instances where not all replicate mean CPUE values reached the CPUE target in the 40 years following the introduction of the Target CPUE HCR (**Table 44**).

These findings reinforce the outcome that in the severely depleted scenarios while the target CPUE HCR can succeed, it can take a long time to successfully rebuild a depleted zone. The use of a TAC limit only exacerbates any delays in recovery and keeps CPUE low.

7.7.4.4 *Maintaining the CPUE Target.*

The target CPUE HCR generally avoids the oscillations in catches and catch rates exhibited by the CPUE gradient HCR. However, longer term oscillations can occur (see section 21.4.8, p.277; **Figure 127** and **Figure 128**). These reflect time-lags in the response of the HCR to changes in catch rates. These come about through the time it takes a change in the TAC to change the amount of spawning biomass and consequent recruitment, and then the time taken to grow to a size past the LML where the abalone join the fishery.

Even though the target CPUE HCR works in all scenarios to attain the selected target, one issue is that in many scenarios it overshoots the target and appears to remain above the specific value selected (see **Figure 129**); this bias in the outcome could be as much as 10 kg/hr higher than the selected target. There appear to be two sources of this bias and both relate to the implementation of the scoring mechanism inside the multi-criterion decision analysis framework. The first problem relates to the need for the requirement for the HCR to operate with scores that are integers. The sum of the integers for each performance measure (only one was used here) determine the overall response of the TAC. A problem arises with the use of a “trunc” mathematical function in the HCR which rounds values

down to the nearest integer. Given the resolution of the HCR in terms of the degree of change imposed on the TAC this by itself tend to bias the outcome high. By rounding values down, and the values often representing units of 10 kg/hr, then CPUE has to rise by a full 10 kg/hr for the score to increment upwards. This implies that the HCR perceives the CPUE to always be lower than it is except when CPUE is exactly at a nice round number. This also means that setting the target at some level which is not a round number (e.g. 85 kg/hr instead of 80 or 90 kg/hr) can also confuse the workings of the HCR and lead to bias. One possible simple solution is to use the mathematical function “round” instead of “trunc”; this has the effect of rounding to the nearest integer, although this can interact badly if the target is does not round to a unit of 10 kg/hr. Further work is required to solve this issue for all circumstances.

The second issue relates to the minimum change permitted by the HCR. Previously small changes are not considered sensible as noisy data may mean the decision would be reversed in the following year, so only relatively large changes are pursued by management. However, enforcing such a restriction automatically sets a limit on the precision with which a target CPUE can be achieved.

As long as this doesn't undermine confidence in the strategy (and HCR) then it is not a major concern, especially as most biases appear to be positive and therefore more conservative. Nevertheless, it would be useful to explore extra options for improving the effectiveness of achieving a set target so as to maximize confidence in the methods. This may not remain an issue once more performance measures are included in the multi-criterion decision analysis framework.

7.7.5 HCR Conclusions

The symmetric CPUE gradient HCR should not be used to provide management advice, although an asymmetric CPUE gradient HCR may prove valuable in a management framework using multiple performance measures. Fortunately, the target CPUE HCR appears to perform to expectations reasonably well. It can recover a depleted stock and can also manage an under-fished stock. Before it is used it would be best to demonstrate a relationship between catches and subsequent catch rates. In addition, the inevitable time-lags between a management response to a given CPUE and the desired effect being exhibited can lead to variable fishery behaviour and unintended over-shoots of catch which in turn lead to variable catch rates and depletion levels. This suggests that even though this HCR operates mostly as intended it could still be improved.

The TAC limit may appear to be a valuable innovation in maintaining a fishery even when the stock status is clearly dire. However, it is potentially a destructive process which can trap a fishery into being perennially over-fished and under-performing. An abalone fishery can become sustainably overfished and stuck with catch rates at which it must become difficult to make a substantial profit.

8 Benefits and Adoption

The national and international review of management objectives and PIs (Stage 1) provided substantial benefit to South Australia by contributing to revision of the Management Plan for that fishery (PIRSA 2012), and the associated harvest strategy. When the Management Plan is revised in 2015, the outputs from the MSE (Stage 2) will be central to the

revision process. In addition, the outputs from the MSE will be used to assist with the design the multi-criterion decision analysis framework to be introduced into the Tasmanian abalone fishery. In Tasmania there are discussion happening now (2013/2014) relating to the most appropriate use of legal minimum lengths combined with appropriate total allowable catches to both maintain sustainability and optimize the use of the resource; there is currently a move to shift to a final LML of 145mm on the south west coast while somewhat reducing the TAC. This stems from the detailed examination of the relationship between LML and TAC in the MSE work leading to an improved appreciation of that relationship by managers and industry. However, as is usually the case, this management option remains highly contentious within industry with strong views expressed both for and against. Critical to the final decision is the timing of the LML change and TAC reductions, particularly in an environment with declining stock levels. The MSE can be particularly helpful during this discussion, by identifying which sequence of changes is optimal, or indeed, if it even matters.

In Western Victoria, the outcomes from the recent application of the new MSE framework to their situation has been used directly in deciding on an appropriate harvest rate (among the range tested) for their current (2014) constant harvest rate management strategy. The harvest rate selected by the participants in the Industry meeting that followed the gathering in which the MSE findings were articulated provided a balance between catches taken combined with a small decrease in the LML, while leading to an sufficiently precautionary rebuilding rate. These decisions were made with an understanding of the range of uncertainties remaining.

These changes have been brought about by on-going interactions and communications with Industry, Managers and other scientists. Numerous meetings and presentations have been made including in Tasmania, Victoria, South Australia, and New South Wales.

9 Further Development

There remain many weaknesses in our understanding of abalone population dynamics and chief among these relates to the recruitment dynamics. The generality and confidence in the outputs of the MSE would be enhanced by further explorations into abalone recruitment processes. This could include a combination of field observations and simulation studies; the latter would be aimed at summarizing what was known and determining whether alternative proposed recruitment mechanisms would constitute sufficient explanation for the variation observed in nature.

There would also be benefit from applying the MSE framework to the full implementation of a multi-criterion decision analysis framework to be used in Tasmania. Within the MCDA this should include testing alternate HCR for a broader range of performance measures and alternative weighting coefficients for the performance measures. A key development issue is addressing the need to include the new spatial performance measures being developed as a result of regulated the full fleet of divers to use GPS data loggers. FRDC project 2013/200 “Testing abalone empirical harvest strategies, for setting TACs and associated LMLs, that include the use of novel spatially explicit performance measures” is designed to do exactly that.

10 Planned Outcomes

A workable set of performance indicators and the quantitative assessment of a suite of Management Strategies through a formal Management Strategy Evaluation Framework will be identified for use in the management of south-eastern fisheries for abalone. These management strategies will be more robust to uncertainty of model structure, biological information, and data availability.

Management advice for abalone fisheries will be improved leading to more opportunities for optimizing the harvest (maximizing the yield and value without compromising the sustainability). The beneficiaries will be the managers and fishers (both commercial and recreational) for abalone stocks.

Develop understanding of the outputs from the MSE among the stakeholders who will use the management strategies identified by the MSE process, by interacting directly with those stakeholders in their own jurisdictions.

Revised management plans, incorporating formal harvest strategies, for SA, and Tas will be facilitated through the findings from application of the MSE to some of the PIs discussed in this report.

11 References

For improved ease of use all references from this report, both the from the front sections and the appendices, have been gathered together in alphabetical order in section 22 at the end of the report.

12 Appendix 1: Intellectual Property

There are no issues concerning intellectual property relating to this research project.

13 Appendix 2: Staff

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14 Appendix 3: Stage 1 – Objectives 1-3

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14.1 Management objectives and performance indicators in abalone fisheries: a review

14.1.1 Introduction

Sustainable management of natural resources is commonly the responsibility of Government. However, co-management approaches, where responsibility for the sustainable utilisation of the resource and maintenance of its biological and ecological relevance is delegated among stakeholders (FRDC 2008) are increasing. Nevertheless, worldwide, sustainable management of fisheries has proven difficult to achieve (McWhinnie 2009), despite intensive efforts (Jenkins 2004). Consequently, evidence of overexploitation and stock collapse is wide-spread (Myers & Worm 2003, Mora *et al.* 2009). Over-fishing has resulted in follow-on effects to ecosystem function and biodiversity (Jackson *et al.* 2001, Lotze *et al.* 2006, Worm *et al.* 2006) and to social and economic status (Maunder *et al.* 2006). Subsequently, many international agreements (UNCLOS 1982, FAO 1995, UNFSA 1995) and national and state legislation (CFA 1991, NSESD 1992, EPBC 1999, NPF 1999, State Fisheries Acts) require fisheries to operate within a management system focused on biological and ecological sustainability. Management systems should (1) avoid overfishing, (2) promote recovery of depleted populations and (3) control the ecosystem effects of fishing. Collectively, these will help to facilitate ecologically sustainable development (ESD) of fisheries.

Modern fisheries management often occurs under a formal framework, such as a fishery management plan (MP). Fishery MPs provide guidance (explicit or implicit) for management decision making, aiming to provide management systems that will achieve ESD. MPs typically identify and describe management objectives for the fishery. Commonly, these include biological (or ecological), economic, governance and social objectives. Assessing performance of the fishery against the objectives described in the MP requires performance indicators (PIs) to be specified for each objective. Assessment of current fishery performance against the objectives can be linked to prescribed management actions (= harvest control rule or decision rule). This approach comprises a ‘harvest strategy’, and is usually the cornerstone of a fishery management plan. MPs with well-developed, robust harvest strategies allow resource stewards to monitor and assess fishery performance, especially its biological and economic standing, and implement actions aimed at matching fishing intensity to stock status (CFHS 2007).

Assessment of fishery performance can only be made against defined criteria through time (**Figure 7**). PIs provide objective criteria against which fishery performance can be assessed, and are a central component of a harvest strategy. To improve functionality, PIs should (1) be directly relevant to each fishery; (2) be estimable with sufficient accuracy to form the basis of clear management actions; (3) reflect the biology of the species concerned and; (4) be agreed to by stakeholders.

Several approaches have been developed to assess fishery performance against PIs. For example, Australian Commonwealth Fisheries are assessed using PIs with clear target and

limit reference points (**Figure 7**; Halliday *et al.* 2001, Caddy 2004, CFHS 2007, Sainsbury 2008). This method is considered necessary to enable Australian export fisheries to meet the requirements of the *Environment Protection and Biodiversity Conservation Act (1999)* (EPBC Act). Reference points are values of a PI "...that can be used as a benchmark of performance against an operational objective" (Fletcher *et al.* 2002). Target reference points (TRP) describe desirable values of PI, and infer a positive position for a fishery. Limit reference points (LRP) describe PI values that are considered unacceptable. Thus, values of LRP identify the lowest level to which fishery performance can drop prior to mandated remedial management action. Trigger reference points are often used with TRP and LRP. They are typically used to initiate a management response (e.g. changing total allowable commercial catch (TACC) or minimum legal size (MLS) limits) before TRP or LRP are reached. Successful implementation of this approach requires (1) stakeholder engagement and acceptance and; (2) documented, prescribed actions should 'worst case' scenarios occur (Caddy 2002).

This approach is not used uniformly across fisheries. Another common approach involves determining whether the PI is above or below some single reference point derived from historical performance (e.g. Tarbath *et al.* 2002, Anon 2007a), or whether the value of the PI has changed significantly through time (Nobes *et al.* 2004). When either of these occur, the PI is identified as having 'triggered'. A common management action in these circumstances is to initiate a review of the circumstances that led to the PI 'triggering'. Consequently, in many cases, these 'triggers' constitute soft LRP.

Obtaining PIs that accurately measure changes in fishery performance is difficult. Thus, despite similarities among many fisheries, particularly those for the same or similar species, different suites of PIs are used to assess fishery performance in different places. Lack of consensus is exacerbated by operational and legislative differences among fisheries, particularly those in different countries and states. These differences mean use of the same set of PIs is not necessarily appropriate, possible, or even desirable.

Abalone (Family: Haliotidae; Genus: Haliotis) are highly prized and valuable marine gastropods inhabiting near-shore reefs (Day & Shepherd 1995) from the shallow subtidal zone to depths around 400 m (Geiger 1999). Historic artisanal abalone fisheries (Buchanan 1985, Guzman del Proo 1992) were followed by large-scale commercial fisheries. Harvesting wild abalone has formed the basis of important commercial, indigenous and recreational fisheries in Alaska (Paul & Paul 1998), Australia (Prince & Shepherd 1992), British Columbia (Donovan & Carefoot 1998), New Zealand (Schiel 1992, Roberts *et al.* 2007), Japan (Sasaki & Shepherd 2001), South Africa (Tarr 1995, Edwards & Plaganyi 2008) and Western North America (Davis 1995, Rogers-Bennett & Butler 2002).

Commercial abalone fisheries developed through the establishment of strong markets in the mid-1970s (Breen 1992), primarily in Asia. Traditional management systems for abalone fisheries included a range of input (limited entry) and output (total allowable commercial catches (TACCs), minimum legal lengths (MLL) and individual, transferable quotas (ITQs)) controls (Breen 1992, Prince & Shepherd 1992). However, strong demand, limited understanding of abalone population spatial structure, disease, habitat loss and illegal fishing (Davis *et al.* 1992, Davis *et al.* 1998, Shepherd *et al.* 1998, Gordon & Cook 2004, Prince 2004, Prince 2005, Branch & Clark 2006, Raemaekers & Britz 2009) have contributed to the rapid decline of the wild abalone catch outside Australasia over the past three decades (Breen 1986, Davis *et al.* 1992, Guzman del Proo 1992, Altstatt *et al.* 1996, Tegner *et al.* 2001, Gordon & Cook 2004, Lessard & Campbell 2007, Hamasaki & Kitada 2008, Raemaekers & Britz 2009).

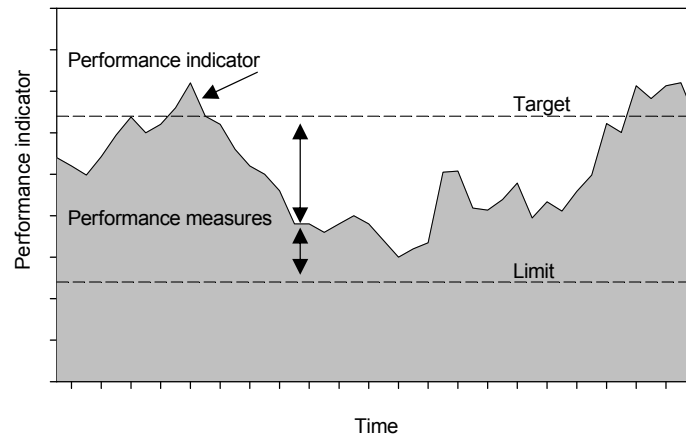


Figure 7. Relationship between a performance indicator, reference points (target and limit) and performance measures (adapted from CFHS 2007).

In general contrast to the broad global trends, Australian abalone fisheries have fared comparatively well, with these fisheries appearing relatively stable and sustainable. Abalone fisheries operate in each of the five most southern states (**Table 2**): Tasmania, Victoria, South Australia (SA), Western Australia (WA) and New South Wales (NSW)). In 2007/08 Australian production of wild abalone was 5,300 t, which was valued at about \$AUD190 million (ABARE 2009). Australian wild caught abalone provides ~50% of global wild catch production (Gordon & Cook 2004). The primary species harvested are blacklip (*H. rubra*; Leach 1815) and greenlip (*H. laevigata*; Donovan 1808), with smaller quantities of Roe's (*H. roei*; Gray 1827) and brownlip (*H. conicopora*; Péron, 1816) also harvested commercially. The Tasmanian wild catch industry is the largest wild abalone fishery in the world, providing around 50% of Australian wild harvest production (Gordon & Cook 2004), and consequently 25% of the annual global harvest. Each state abalone fishery is comprised of a number of spatial units (Fishing Zones, Areas or Regions) against which TACCs are allocated.

Fishing histories and management arrangements vary among states, and have evolved over a period of ~50 years. However, they commonly include a range of input (e.g. limited entry) and output (e.g. MLL) and spatially-managed (Zonal) individually transferable quota (ITQ) controls, underpinned by fishery assessments based on a broad range of fishery-dependent and fishery-independent data and guided by formal management plans (e.g. Management Plan for the SA Abalone Fishery (Nobes *et al.* 2004); Tasmanian Abalone Fishery Revised Policy Paper (DPIWE 2000), NSW Abalone Fishery Management Strategy (Anon 2007a); Victorian Abalone Fishery Management Plan (Anon 2002), WA Integrated Fishery Management Report – Abalone Resource (Anon 2005)). These management plans aim to ensure future sustainability of these fisheries through contemporary management approaches.

A key feature of management plans in recent years, partly in response to changes in biodiversity conservation legislation, has been the development of management objectives assessed against PIs, and, in rare cases, associated TRPs and LRPs. As elsewhere, these PIs collectively inform the decisions upon which management of the fishery is largely de-

pendent. To ensure appropriate management and, consequently, future sustainability of these fisheries, these PIs must be robust. Thus, for biological PIs, they must provide clear and timely indications of variation in abalone abundance and/or population structure. Hence, they must be sensitive to and effective at detecting change. Without this capability, they will fail to identify Zones/Regions/Reefs where the resource may be heading towards being overfished, or, alternatively, where the resource could sustain additional fishing pressure. Consequently, there is a strong need for spatially-relevant, defensible assessments for abalone that have a predictive capacity. An additional complication is the desirable need to use agreed PIs to provide a single, unambiguous statement of biological stock status to fishery managers. In fisheries where multiple PIs are used, their relative importance is not described and, when their values provide conflicting assessment of fishery performance, this is especially challenging.

While a range of PIs for finfish fisheries are well accepted as tools for fishery management (Caddy 1998), similar levels of agreement have not been reached for abalone fisheries. Consequently, numerous PIs are used in the management of abalone fisheries in Australia (Gorfine *et al.* 2001). The utility of the multitude of PIs used in the management of Australian abalone fisheries to act either as an ‘early warning signal’ or as an indicator of improving resource status for these fisheries is poorly understood. The uncertainty in the quality of the PIs has also limited development of formal, harvest-control rules that link PI-based assessments of stock status with clearly-defined and prescribed management outcomes. The absence of formal connections between PIs and management responses means harvest strategies for these fisheries are incomplete.

Overarching the lack of consistency in approach among states, the poor understanding of the applicability of the PIs used and the lack of formal control rules, is the topical issue of aligning the scale of fishery assessment and management with the scale of biological stocks (Mayfield & Saunders 2008, Prince *et al.* 2008). Recent studies have confirmed the high level of independence of abalone populations (e.g. Temby *et al.* 2007, Miller *et al.* 2009), which indicates that future abalone fishery assessments need to occur at finer spatial scales and suggests that consequent management might need to be at finer scales (e.g. sub-zones, fishing areas, mapcodes or reefs) than that commonly employed (i.e. zone, region). This recognition of the over-riding importance of spatial structure in abalone populations also necessitates reconsideration of fishery performance measures

Table 2. Summary of the structure of Australian state-based abalone fisheries, including the number of fishing zones, species caught, quota holders, licences or shareholders, total allowable commercial catch (TACC) and total value of the fishery. BL: Blacklip (*H. rubra*); GL: Greenlip (*H. laevigata*); BR: Brownlip (*H. conicopora*); R: Roe's abalone (*H. roei*).

State	Fishing Zones (with independent TACCs)	Species	No. of fishing licences, quota holders or shareholder arrangements	TACC (t)	Total value (\$ M) 2007/08 ^a
Tasmania				2604 ¹	94.57
	Eastern	BL	320 quota holders endorse 125 fishing licences to catch allocated TACC ^b .	850.5 (2009)	
	Western	BL		924 (2009)	
	Central Western	BL		304.5 (2009)	
	Northern	BL		332.5 (2009)	
	Bass Strait	BL		70 (2009)	
	Greenlip fishery	GL		122.5 (2009)	
Victoria				918.2 ²	43.95
	Eastern Zone	BL	23 fishing licences	460 (2010)	
	Central Zone	BL & GL	34 fishing licences	429 (BL, 2010) & 3.4 (GL, 2010)	
	Western Zone	BL & GL	14 fishing licences	16 (BL, 2010) & 9.8 (GL, 2010)	
South Australia				886.24 ³	36.19
	Western Zone	- Region A	BL & GL	23 fishing licences	293.25 (BL, 2009) & 227.7 (GL, 2009)
		- Region B	BL & GL	23 fishing licences	
	Central Zone		BL & GL	6 fishing licences	143.1 (GL, 2009) & 24.3 (BL, 2009)
		- Cowell	GL	6 fishing licences	
	Southern Zone	- non FDA ^c	BL	6 fishing licences	99 (2009)
		- FDA	BL	6 fishing licences	45 (2009)
		- FDA & non-FDA	GL	6 fishing licences	6 (2009)
Western Australia				314.5 ⁴	10.17
	Area 1	GL, BR & R	20 fishing licences	3.2 (GL), 0.1 (BR) & 9.9 (R)	
	Area 2	GL, BR & R	21 fishing licences	74.7 (GL), 23.2 (BR) & 19.8 (R)	
	Area 3	GL & BR	8 fishing licences	85.3 (GL), 21.3 (BR)	
	Area 5	R	21 fishing licences	20	
	Area 6	R	10 fishing licences	12	
	Area 7	R	13 fishing licences	36	
	Area 8	R	12 fishing licences	9	
New South Wales				109.4 ^{5,†}	
	Region 1	BL	46 shareholders with 3454 shares (39 shareholders with ≥70 shares ^d .)	0.62 [†]	
	Region 2	BL		2.40 [†]	
	Region 3	BL		14.15 [†]	
	Region 4	BL		24.13 [†]	
	Region 5	BL		23.31 [†]	
	Region 6	BL		44.78 [†]	

¹ (Anon 2008a); ² (Anon 2010); ³ (Mayfield *et al.* 2008a, Chick *et al.* 2009); ⁴ (Hart *et al.* 2009a); ⁵ (Anon 2008c); ^a (ABARE 2009); ^b 35 00 quota units among 320 quota owners (1 unit equates to ~744kg of TACC); ^c FDA: Fishdown area (access to stock below MLL); ^d ≥70 shares to harvest the apportioned TACC. [†] Reported catch – not TACC

The objectives of this section are to (1) review the fishery management objectives and PIs used in Australian abalone fisheries, and similar fisheries elsewhere, and (2) generate a ‘working list’ of prospective PIs as the basis for subsequent analyses described in later sections. To do this, we first document and describe the management objectives (Section 2.2.1) and PIs currently ‘prescribed’ for each Australian state-based abalone fishery (Section 2.2.2). We note that the SA, Victorian and Tasmanian management plans are currently being reviewed or developed, and that the information presented here may change through time. As management arrangements change through time, current assessments may differ substantially from those that are ‘prescribed’. Hence, informal ‘indicators’ of fishery performance, that are used in Australian state abalone fishery assessment reports to aid evaluation of stock status are described and potential new fishery PIs, particularly those under development or consideration are identified (Sections 2.2.3 and 2.2.4, respectively). Management objectives and PIs used in abalone fisheries elsewhere are summarized in Section 2.3, and those for dive fisheries based on other commercially important invertebrate species are outlined in Section 2.4. The discussion synthesizes the information presented in previous sections, and provides a comprehensive ‘working list’ of prospective PIs that forms the basis for studies in subsequent sections.

14.1.2 Management objectives and PIs in Australian abalone fisheries

Information presented in this section was obtained from Australian abalone fishery MPs, other documents describing Australian, state-based abalone fishery management systems, and state-based, fishery assessment reports. Management systems are prescribed for each abalone fishery, but approaches vary among states. These approaches range from a combination of legislative, statutory and policy documents (Tasmania and WA) to single MPs with stipulated management objectives, PIs and trigger points (Victoria, SA and NSW).

14.1.3 Management objectives

Fisheries management requires a fine balance to achieve multiple, potentially-competing objectives. Well developed objectives provide guidance for making management decisions that comply with legislative requirements for ESD. Management objectives among the Australian state-based abalone fisheries are commonly divided into biological (includes ecological and environmental), economic, governance (management), and social categories (**Table 3**). Whilst each objective should be considered in formulating management action, the biological objectives provide the principal management direction, as required by the EPBC Act and *Guidelines for the Ecologically Sustainable Management of Fisheries* (DoEWR 2007).

There are substantial differences among states regarding the specificity, diversity and number of abalone fishery management objectives (**Table 3**). These objectives are also typically general in nature, seldom specific and at times contradictory. Consequently, interpreting their specific intent is often difficult. This low level of specificity almost certainly diminishes their functionality, which can lead to uncertainty in decisions aimed at driving the fishery towards its objectives. Similar objectives have also been placed into different categories: reducing illegal harvest is a social objective in SA, but a governance objective in Victoria and NSW.

All states prescribe biological and economic objectives. These reflect the (1) importance of ensuring the stocks are fished sustainably and within an ESD framework, and (2) high value of the product, licences and quota. Tasmania, Victoria, SA and NSW each have social objectives, and governance objectives are prescribed for Victoria, NSW and WA. The

Victorian (14) and SA (12) fisheries have the greatest number of specified objectives, with Tasmania (5) and WA (4) having the least.

The highest level of consistency among states is evident in the biological objectives (**Table 3**). This commonality is understandable, as the fishery in each state operates within the same national and similar state-based requirements for biological and ecological sustainability. Nevertheless, some differences among states remain, and are predominately related to the specificity of the biological objectives. For example, objectives in Tasmania and Victoria require sustainable development of natural resources and sustained productive capacity, respectively, without reference to a mechanism to ensure sustainable fishing. However, both these states specify maintenance of abalone genetic diversity and integrity within their biological objectives. In contrast, objectives relating to sustainability in SA, WA and NSW are more specific and focus on maintaining recruitment through the maintenance or increase in egg production (SA), breeding stock (WA) and spawning biomass (NSW). Each state also has biological objectives regarding the broader ecosystem. Again, these differ among states, ranging from ‘maintain ecological processes’ (Tasmania), ‘ecosystem health’ (Victoria), ‘habitat maintenance’ (WA), ‘minimising the environmental impacts of fishing’ (SA) and ‘conservation of biological diversity’ (NSW). Whilst PIs for assessing these fisheries against the sustainability objectives are generally well developed, only Victoria specifies a PI for assessing the fishery against the ecological objectives (see Section 2.2.2).

Consistency in the economic objectives among states reflects the unit value of abalone (~AUD 30.kg⁻¹), the high asset value of fishing licences, dive entitlements and quota, and the large economic returns to local communities and state economies. The focus in all states is on economic development, viability and return on investment (**Table 3**). Whilst economic efficiency and cost reduction are implicit within these components, they are explicitly identified only for Victoria and SA. As with the biological objectives, none of the economic objectives specify the mechanism for achieving that objective. However, with the exception of Tasmania, each state has specified PIs for assessing fishery performance against the economic objectives (see Section 2.2.2.2).

Governance and social objectives are less similar among states when compared to the biological and economic objectives. For example, no governance objectives are identified for SA or Tasmania, and no social objectives are prescribed for WA. Governance objectives in Victoria, WA and NSW outline goals of cost-effective, efficient and transparent management practices, and facilitation of shared management responsibility. Additional governance objectives for NSW and Victoria relate to minimizing illegal fishing. While PIs for assessing fishery performance against the governance (see Section 2.2.2.3) objectives are established, they tend to be less well defined than those for the biological and economic objectives.

Social objectives are identified for the Tasmanian, Victorian, SA and NSW abalone fisheries. These objectives generally focus on (1) equitable stakeholder (recreational, indigenous and commercial) access to the resource (Victoria, SA and NSW) and (2) shared responsibility for management of the abalone stocks (Tasmania and NSW). ‘Appropriate community returns’ and ‘adequate compliance resources’ form additional social objectives for Victoria and SA, respectively. The PIs for assessing fishery performance against these social objectives (see Section 2.2.2.4) are less well defined than those for the biological, economic and governance objectives.

Table 3 .Management objectives of Australian State Abalone Fisheries. Ref No. provides a reference for each of these objectives described in other Tables.

STATE	FISHERY CATEGORY	Ref. No.
Biological (Ecological and/or Environmental)		
Tasmania ¹	<ul style="list-style-type: none"> to promote the sustainable development of natural and physical resources and the maintenance of ecological processes and genetic diversity (Tas. Schedule 1, Objective a.) 	T1
Victoria ²	<ul style="list-style-type: none"> Productive capacity of stocks sustained into the future at low levels of risk. Ecosystem health (including genetic integrity of abalone) not jeopardised by abalone fishery practices Management responsive to changes in ecosystem health 	V1 V2 V3
South Australia ³	<ul style="list-style-type: none"> Control, measure and regulate all catches/extractions from the resource Monitor and control disease To minimise the environmental impacts of abalone fishing and promote conservation measures Maintain sufficient egg and sperm production to provide for adequate levels of recruitment 	S1 S2 S3 S4
Western Australia ⁴	<ul style="list-style-type: none"> Maintain sustainability of the State's abalone stocks through maintenance of the breeding stock and habitat 	W1
New South Wales ⁵	<ul style="list-style-type: none"> Manage commercial harvesting of abalone to promote the conservation of biological diversity in the coastal environment (NSW Plan, Goal 1) <ul style="list-style-type: none"> Increase knowledge and minimise any adverse impacts of harvesting abalone on bycatch, associated habitats and ecosystems Maintain or rebuild the biomass of abalone to ensure stock sustainability (NSW Plan, Goal 2) <ul style="list-style-type: none"> To maintain or increase the spawning and exploitable biomass of abalone To improve the efficiency of harvesting and investigate the potential of techniques to rebuild populations of abalone To address impacts from factors external to the commercial Abalone Fishery 	N1 N1a N2 N2a N2b N2c
Economic		
Tasmania ¹	<ul style="list-style-type: none"> to facilitate economic development in accordance with the objectives a, b and c (Tas. Schedule 1, Objective d; see Biological and Social objectives) 	T2
Victoria ²	<ul style="list-style-type: none"> Opportunities for commercial production fully utilised Economically efficient commercial production Commercial production directly enhanced through appropriate activities (including aquaculture, ranching, resource enhancement, and access to stunted stocks) 	V9 V10 V11
South Australia ³	<ul style="list-style-type: none"> To maintain the abalone fishery at a level that provides for fair and reasonable economic benefits to licence holders To recover an economic return sufficient to cover costs To provide for economic efficiency and flexibility in management arrangements Develop harvest strategies that minimise costs Optimise yield from the fishery 	S5 S6 S7 S8 S9
Western Australia ⁴	<ul style="list-style-type: none"> Maximise the economic return from the abalone resource to the community while maintaining sustainability of the stock and the habitat 	W4
New South Wales ⁵	<ul style="list-style-type: none"> To promote the economic viability of the fishery (NSW Plan, Goal 4) <ul style="list-style-type: none"> To promote the long term economic viability of commercial abalone fishing 	N6 N6a
Governance (Management)		
Victoria ²	<ul style="list-style-type: none"> Management which is cost-effective and transparent Recovery of the attributable costs of management, including research and compliance Stakeholders and government fisheries administration sharing responsibility and involvement in management Compliance targets for licensed sectors of the fishery achieved and monitored Illegal activities prevented and targets for reduction of theft monitored and achieved 	V4 V5 V6 V7 V8
Western Australia ⁴	<ul style="list-style-type: none"> Ensure cost-effective management of the fishery, with special reference to developing systems in collaboration with clients to fully identify all costs of running the sub-program for cost recovery Encourage maximum commercial flexibility and administrative simplicity from industry participants 	W2 W3
New South Wales ⁵	<ul style="list-style-type: none"> Facilitate effective management arrangements and provision of an efficient fisheries management service (NSW Plan, Goal 3) <ul style="list-style-type: none"> To facilitate the delivery of effective and efficient management services Facilitate appropriate research and monitoring of the Abalone Fishery (NSW Plan, Goal 6) <ul style="list-style-type: none"> To collect information on the Abalone Fishery and the environment on which it operates in a timely manner To keep informed of research and management initiatives in other jurisdictions Achieve a high level of compliance within the Abalone Fishery (NSW Plan, Goal 7) <ul style="list-style-type: none"> Promote a high level of compliance in the fishery To continue to minimise the illegal catch of abalone 	N3 N3a N4 N4a N4b N5 N5a N5b
Social		
Tasmania ¹	<ul style="list-style-type: none"> to provide for the fair, orderly and sustainable use and development of air, land and water (Tas. Schedule 1, Objective b.) to encourage public involvement in resource management and planning (Tas. Schedule 1, Objective c.) to promote the sharing of responsibility for resource management and planning between the different spheres of Government, the community and industry in the State (Tas. Schedule 1, Objective e) 	T3 T4 T5
Victoria ²	<ul style="list-style-type: none"> Equitable assignment of productive capacity between commercial, recreational, indigenous, and non-exploitative uses Recognition of past access by indigenous Australians Appropriate community return where there is commercial use of publicly owned abalone stocks and habitats 	V12 V13 V14
South Australia ³	<ul style="list-style-type: none"> To maintain and provide for reasonable access to abalone resources for recreational fishers To provide for adequate compliance resources Improve access, for the community, to information on the abalone fishery 	S10 S11 S12
New South Wales ⁵	<ul style="list-style-type: none"> To appropriately share the resource and harvest abalone in a manner that minimises negative social and economic impacts (NSW Plan, Goal 5) <ul style="list-style-type: none"> Mitigate negative impacts of the Abalone Fishery on Aboriginal cultural heritage To minimise any negative impacts of the Abalone Fishery on Aboriginal and European cultural items in the vicinity of abalone harvesting areas Ensure adequate stakeholder involvement and community consultation (NSW Plan, Goal 8) <ul style="list-style-type: none"> To ensure the Abalone Management Advisory Committee communicates effectively with shareholders, other industry sectors and other stakeholders To promote community awareness about the importance of habitat and other environmental factors that affect abalone 	N7 N7a N7b N8 N8a N8b

¹. *Living Marine Resources Management Act 1995. Schedule 1 Objectives of the Resource Management and Planning System of Tasmania*; ². *Victorian Abalone Fishery Management Plan*; ³. *Management Plan for the South Australian Abalone Fishery*; ⁴. *Integrated Fisheries Management Report – Abalone Fishery. Fisheries Management Paper No.204*; ⁵. *Fishery Management Strategy for the NSW Abalone Fishery*.

14.1.3.1 *'Formal' PIs*

Informative and robust PIs are required to objectively assess fishery performance against specified management objectives and guide management actions, such as changes to harvest rules (e.g. increases or decreases in TACCs), to ensure these objectives are met. The identification of appropriate indicators is also essential to ensure management actions, based on their status, are confidently supported by stakeholders.

A diverse range of PIs are prescribed for the assessment of Australian state-based abalone fisheries (**Table 4**). In **Table 4**, PIs are grouped within management objective categories (i.e. biological, economic, governance and social), then by source of the data from which they are measured (i.e. fishery-dependent, fishery-independent, etc.). Where available, the specific measure of the indicator and the trigger point are detailed (denoted by a superscript for each state). The spatial scale and state of application are also identified and management objectives against which each PI is assigned is cross referenced against those identified in **Table 3**. Where harvest control rules follow the 'triggering' of PIs, these are described in the text.

14.1.3.2 *PIs for assessment against biological objectives*

Overwhelmingly, most PIs relate to assessing fishery performance against biological objectives and, almost exclusively, those relating to sustainability rather than ecosystem integrity. The PIs are obtained from a broad range of sources including fishery-dependent and fishery-independent data and outputs from numerical models. Additional PIs are determined from less quantitative information provided by divers, licence holders and other stakeholders in the fisheries (**Table 4**).

14.1.3.3 *Indicators obtained from fishery-dependent data*

Most of the PIs for assessing performance of each state-based abalone fishery against the biological objectives are obtained from fishery-dependent data. These fishery-dependent data describe measures of the target species, including catch (by number and by weight), fishing location, effort, measures of catch rate and weight/length-frequency distribution of the catch.

Catch and spatial distribution of the catch – TACCs are the primary output control used to manage Australian abalone fisheries. Consequently, PIs based on catch are commonly used to assess fishery performance against biological objectives. Nevertheless, different approaches are used in each state. These approaches include change in total catch, catch as a percent of TACC, catch in relation to fishing history and change in the spatial distribution of catch. SA and WA use catch as a percentage of the TACC as a key performance indicator. This measure provides information on the capacity of (1) stocks to support the TACC and, (2) the fleet to harvest the TACC. The trigger in SA is when <90% of the TACC is harvested. For WA, the trigger occurs when total catch exceeds the TACC. In Tasmania, fishing areas that produce large catches or areas where catch is significantly different to previous years are prescribed key areas for assessment. Changes in the spatial distribution of the catch are also used in SA to monitor changes in the relative performance of individual fishing areas, because abalone fisheries are vulnerable to serial depletion.

Victoria, WA and NSW apply catch related indices at small spatial scales. This is primarily achieved by examining the relationship between the observed (C_O) and expected (C_E) catch for each spatial unit. For NSW and WA, the trigger is when $C_O > C_E$. In Victoria, the trigger is when C_O falls outside the reference range for C_E (i.e. less than the minimum C_E ; more than the maximum C_E), which is designed to reflect the long-term productivity of each spatial unit.

Effort – PIs based on fishing effort are rarely used to assess fishery performance. In WA, the range of expected diver days (D_E) is specified annually, with this PI triggering if the actual number of diver days falls outside this range. SA uses two effort-based PIs: total effort and mean daily effort (MDE). The total effort PI is applied at the Zonal scale, with the trigger being a statistically significant difference in total effort over a five year period. MDE is applied at a smaller spatial scale, Fishing Area, from which a minimum percentage of the TACC has been harvested. This measure is determined as the average number of hours fished on each fishing day (hr.day^{-1}). The triggers for MDE are a statistically significant difference (1) between years and (2) a trend statistically different from zero over a five year period.

Catch rate – Catch rates are one of the most common PIs by which Australian state-based abalone fisheries are assessed against biological objectives. These measures of stock abundance are determined from the catch and effort data, and thus comprise derived PIs for assessing stock status. Two PIs are typically employed: mean daily catch (kg.day^{-1} ; MDC) and mean catch per hour (kg.hr^{-1} ; CPUE). MDC is used in SA and WA. In SA, MDC is estimated for each fishing area from which a minimum percentage of the TACC has been harvested. The triggers are the same as those for MDE (see above). For WA, an expected range of MDC is determined, again for each area. If the observed MDC exceeds this range it triggers the PI.

Tasmania and SA use CPUE for assessing stock status. In SA, CPUE is determined and assessed as for MDC. In Tasmania, CPUE is calculated as the geometric, rather than the arithmetic mean and is also estimated for each spatial unit (fishing block), from which catch is significantly different to previous levels. The PI is triggered when the CPUE falls below a range defined from a historical reference period. WA is the only state to prescribe standardised CPUE (sCPUE) as a PI. Their approach is similar to that for Commonwealth fisheries, in that spatially-relevant, TRP and LRP have been derived from a reference year. In some areas of the WA fishery, this PI constitutes the sole measure of fishery performance, from which management actions (i.e. harvest control rules) are mandated. In other WA areas, sCPUE is integrated with estimates of fishing mortality (F) prior to the harvest control rules being applied.

Table 4. Summary of ‘formal’ PIs used to assess Australian abalone fishery management objectives. Scale: Large (L) - entire fishery; Medium (M) - Zone/Region; Small (S) - smallest management unit.

Management Objective Category	Data source	Performance Indicator (general description)	Performance Indicator (specific measure)	Trigger	Scale	State	Management Objective Ref No. (see Table 2)
Biological	Fishery-dependent	Catch	Percent of TACC harvested	<90% of TACC ³ ; TACC exceeded ^{4a}	L	SA; WA;	S1, W1
			Percent of expected catch	Annual catch at extreme of range Ref ₁₉₈₈₋₂₀₀₀ ² ; Catch exceeds Area/Regional target ^{4a, 5} .	M, S	Vic, WA, NSW	V1, W1, N2
			Spatial distribution of catch	No trigger ^{1b} ; Change in rank order of top 5 ³ .	L	Tas, SA	T1, S1
		Effort	Diver days	Range specified annually ^{4a}	L	WA	W1
			Total hours	5 year trend ³	L	SA	S1
			hr.day ⁻¹	2 & 5 year trend from areas >5 or 10% TACC ³ .	S	SA	S1
		Catch rate (CPUE)	kg.day ⁻¹	Whole weight - 2 & 5 year trend from areas >5-15% TACC ³ ; Meat weight range per Area ^{4a}	M, S	SA; WA	S1, W1
			kg.day ⁻¹ (ratio species)	2 & 5 year trend from areas >15% TACC ³ .	M, S	SA	S1
			kg.hr ⁻¹	95% or 75% of lowest Ref ₁₉₉₂₋₅ ^{1b} ; 2 & 5 year trend areas >5-15% TACC ³ .	L, M, S	Tas.; SA	T1, S1
			kg.hr ⁻¹ (standardised)	falls outside Target or Limit Ref _{Year} (dependent on Species and Area) ^{4b} .	L, M, S	WA	W1
	Length structure of the catch	Median length	From time series where >4% of catch measured ^{4b} .	M, S	Tas	T1	
		Mean length	2 & 5 year trend from areas where a fixed % TACC harvested ³ ; No trigger ^{4a} .	M, S	SA, WA	S1, W1	
	Fishing mortality (F)	F.yr ⁻¹	not specified - based on growth and length structure of commercial catch ^{4a} .	S	WA	W1	
	Fishery-independent	Density	Legal no.m ⁻²	2 & 5 year trend from survey area ³ .	S	SA	S1,
			Sub-legal no.m ⁻²	2 & 5 year trend from survey area ³ .	S	SA	S1,
			Mature no. m ⁻²	2 & 5 year trend from survey area ³ ; Outside range (28-34m ²) Area 7 - <i>H. roei</i> only ^{4a} .	S	SA, WA	S4, W1
			Total no. m ⁻²	2 & 5 year trend from survey area - Southern Zone GL only ³ ; no trigger ^{4a} .	S	SA, WA	S1, W1
			Species ratio	2 & 5 year trend from survey area - Southern Zone GL only ³ .	S	SA	S1,
			Immature no. m ⁻²	Outside range (28-38m ²) Area 7 - <i>H. roei</i> only ^{4a} .	S	WA	W1
	Model outputs	Biomass (population model)	Mature	90% B ₂₀₀₀ 80% confidence ² ; 15%<Ref ₁₉₉₄ ⁵ ; >50% of 15%<Ref ₁₉₉₄ in 5 years ⁵ .	L, M ⁵	Vic., NSW	V1, N2a
Legal (exploitable)			15%<Ref ₁₉₉₄ ⁵ ; >50% of 15%<Ref ₁₉₉₄ in 5 years ⁵ .	L, M ⁵	NSW	N2a	
	Egg production	Percent of unfished (virgin)	no trigger ^{1b} ; <50% of unfished, from areas >5% TACC or key areas ³ and Areas 2 & 3 ^{4c} .	S	Tas.; SA; WA	T1, S4, W1	
Other data sources							
Industry	Diver assessment	Annual report	Change in stock status from areas >5 or 10% TACC per Zone respectively ³ .	M	SA	S1	
		Illegal catch	Report produced to AFMC	No report	L	SA	S1
Recreational	Recreational catch	No. prosecutions	2 & 5 year trend from fishery ³ ;	L	SA	S1	
		Report produced to AFMC	No report	L	SA	S1	
Environmental	Monitor & report disease impacts	Disease presence & spread	disease reported outside existing areas ³ ; Pest & disease management not followed ⁵	L	SA, NSW	S2, N2	
			Commercial harvest discard report not provided ¹ .	L	SA	S1	
		Change in diseased catch from (1) Zone >10% (2 years) and (2) individuals >25% (2 years) ³ .	M, S	SA	S2		
	Annual survey of harvesting technique	Harvest method	Harvesting by means other than abalone iron ³ .	M	SA	S3	
		Information available to consider MPA	Environmental information	No environmental information ³	L	SA	S3
	Research proposal for species interactions	Proposal developed	No proposal ¹ .	L	SA	S3	
	Ecosystem effects of fishing	Impact of fishing on ecosystem	Indicies at 90% of average Ref 3 years (lower limit) ² ; demonstrable impact of fishing ⁵ .	L	Vic, NSW	V2, V3, N1a	
	Administrative	Commercial fishery data	Percent received	<100% ³ ; <90% ⁵ received within appropriate time	L, M	SA, NSW	S1, N2
		Catch and Effort database	Percent received and entered	<100% ³ ; <90% ⁵ logbook data received and entered	L, M	SA, NSW	S1, N2,
		Performance of Service Level Agreement	Contractual agreements met	Fail to deliver reports ³ .	L	SA	S1
Adherence to Industry Code of Practice		Breaches of CoP	Unacceptable level of breaches of CoP ⁵ .	L	NSW	N1a	

Table 3. (continued).

Management objective category	Data source	Performance Indicator (general description)	Performance Indicator (specific measure)	Trigger	Scale	State	Management Objective Ref No. (see Table 2)	
Economic	Fishery-dependent	Catch	Percent of TACC harvested	<85% of TACC ⁵	L	NSW	N6	
		Effort	Total hours	2 year trend >25% ³	L	SA	S8	
		Catch rate (CPUE)	kg.hr ⁻¹	No trigger ² ; 85% Ref ₁₉₉₄ ⁵	L	Vic., NSW	V10, N6	
	Other data sources	Management costs/fees	GVP ³ ; Fees	>10% GVP ³ ; >CPI for 2 years ³	L	SA, NSW	S6, N6	
		Commercial yield	Yield & capacity ² ; YPR (\$) ³	No Ref point ² ; YPR not maximised ³	L	Vic., SA	V9, V11, S9	
		TACC allocated to Licences as ITQs		Fail to deliver	L	SA	S7	
		Gross Value of Product (GVP)		Negative 5 year trend	L	SA	S5	
		Operating costs		Labour, Lic. Fees and Maintenance costs increase >10% 2 years	L	SA	S8	
		Fishery profit		Fishery profit at 70% Ref _{3 years} (lower limit)	L	Vic.	V10	
		Return on capital		no data	L	SA	S8	
		Beach price		85% Ref _{2003.04} (capacity to pay) i.e. <\$35.7kg ⁻¹	L	NSW	N6	
		Number of divers		Number of divers is outside range determined by NSW DPI	L	NSW	N6	
		Governance	Management costs/fees	Total costs ²	No trigger ²	L	Vic.	V4
			Attributable cost recovery		No trigger ²	L	Vic.	V5
Scientific data available	Data to inform TACC		Inadequate data (determined by TAC committee) ⁵	L	NSW	N4		
Performance of Service Level Agreement	Contractual agreements met		Unsatisfactory delivery ⁵	L	NSW	N3		
Review of research, management & compliance plans	Plan review		Plan expire without review	L	NSW	N4, N5		
Licensed sector compliance (Comm. & Rec.)	Compliance indices		No trigger ²	L	Vic.	V7		
Illegal activity	Compliance indices		No trigger ²	L	Vic.	V7		
Commercial compliance (fishing)	Percent minor & serious offences		>20% minor offences; 10% serious offences	L	NSW	N5a		
Commercial compliance (non-fishing)	Percent minor & serious offences		>20% minor offences; 10% serious offences	L	NSW	N5a		
Co-management	Existence/function of entities		No trigger ²	L	Vic.	V6		
Social	Catch distribution among sectors			>25% difference between commercial and non-commercial catch each 5 years ⁵	L	NSW	N7	
	recreational kg.yr ⁻¹		25% increase over 3 years ³ ;	L	SA	S10		
	Indigenous access		No trigger ²	L	Vic.	V13		
	Adherence to Industry Code of Practice	Breaches of CoP	Non-compliance ³ ; Unacceptable level of breaches of CoP ⁵	L	SA, NSW	S11, N7, N8		
	Licensed sector compliance (Comm. & Rec.)	Compliance indices	No. prosecutions >3 year average ³	L	SA	S11		
	Annual risk assessment		No trigger ³	L	SA	S11		
	Productive capacity of sectors	Yield ratios	No trigger ²	L	Vic.	V12		
	Community cost of resource use	Community return (royalty)	No trigger ²	L	Vic.	V14		
	Consultation	ABMAC and Port meetings	<2 ABMAC fin. Year ¹ , <1 Port meeting	L	NSW	N8		
	Participate in education and awareness programs		No participation	L	SA	S12		
	Status and use of web site		No use of web site	L	SA	S12		
	Volunteer membership of Fishcare program		Not represented in all regions	L, M	SA	S12		

^{1a}. The Tasmanian Abalone Fishery Revised Policy Paper (DPIWE 2000); ^{1b}. Fishery Assessment Report - Tasmanian Abalone Fishery 2008 (Tarbath & Gardner 2009); ². Victorian Abalone Fishery Management Plan; ³. Management Plan for the South Australian Abalone Fishery (Nobes et al.2004) ; ^{4a}. Integrated fisheries management report - Abalone resource. WA Fisheries Management Paper No. 204; ^{4b}. Performance indicators, biological reference points and decision rules for Western Australian abalone fisheries (*Haliotis* sp.): (1) Standardised catch per unit effort (Hart et al. 2009); ^{4c}. (Hart et al. 2010); ⁵. Fishery Management Strategy for the NSW Abalone Fishery; ⁶. TACC Management Report - Abalone 2008.

Length structure of the commercial catch – Different measures of the length structure of the commercial catch are used as PIs to assess fishery performance against biological objectives in the Tasmanian, SA and WA abalone fisheries. In Tasmania, median lengths are estimated for fishing areas from which $>4\%$ of the catch has been measured. No trigger points are described for this measure in the Tasmanian abalone fishery. However, changes in median length are interpreted, along with changes in the 25th and 75th percentiles and trends in CPUE, to infer changes in levels of fishing pressure and/recruitment. In SA and WA, the mean length of the commercial catch comprises the PI. In SA, mean length is estimated from data available for those fishing areas from which a minimum percentage of the TACC has been harvested. The trigger for this performance indicator is the same as that described for MDE (see above). In WA, mean length is estimated for each fishing area, but no trigger has yet been assigned to this PI.

Fishing mortality – F is identified as a biological PI in WA, and is estimated using a length-based, catch-curve analysis (Hart *et al.* 2010). Estimates of F and sCPUE are combined in a risk assessment framework from which harvest control rules (i.e. $\geq 30\%$ TACC decrease, 10% TACC decrease, TACC unchanged or $\geq 10\%$ TACC increase) are implemented.

14.1.3.4 *Indicators obtained from fishery-independent data*

Although fishery-independent survey measures of abalone abundance have been undertaken in SA, Victoria, NSW and WA, outputs from these surveys are only prescribed as PIs in SA and WA. In SA, three PIs are estimated from survey data and used to assess fishery performance against biological objectives. These are the relative abundance of legal-sized, sub-legal-sized and mature (\geq size at first maturity, L_{50}) abalone. The PIs are calculated for each survey site, with the triggers determined in the same way as those for MDE (i.e. a statistically significant difference (1) between years and (2) a trend statistically different from zero over a five year period). In the Southern Zone Abalone Fishery of SA, fishery-independent, survey-based PIs for greenlip abalone are based on (1) total density, and (2) the ratio between greenlip and blacklip abalone at each survey location. This is because greenlip abalone are relatively less abundant in this Zone. In WA, the PIs are the density of mature and immature *H. roei* from surveys conducted in Area 7. Formal reference points have been defined for these PIs, but there is no associated control rule defining subsequent management action. Survey estimates of greenlip and brownlip abalone density are also described as PIs for Areas 2 and 3, but no triggers have yet been assigned to these PIs.

14.1.3.5 *Indicators obtained from model outputs*

Several PIs for assessing fishery performance against biological objectives are obtained from the outputs of two numerical models. The first, an integrated, length-structured, Bayesian, stock-assessment (population dynamic) model, provides numerous outputs including estimates of mature, legal-sized and sub-legal-sized biomass, fishing mortality (F), natural mortality (M), exploitation rate and recruitment (R). The second models used are simpler, egg-per-recruit models, from which estimates of percent retained egg production (PREP) are the primary output (**Table 4**).

Biomass – Length-structured, stock-assessment models are used in the Victorian and NSW abalone fisheries. For both states, the principal model-based PI is the estimate of mature biomass (B_M). Modelling is undertaken, and the PI applied, at a zonal scale in Victoria, and at a fishery-wide and regional scale in NSW. In Victoria, this PI is triggered when there is an 80%

probability that the estimate of B_M is <90% of that in 2000 (the specified reference year, denoted B_{2000}). For NSW, the B_M PI triggers when B_M is >15% below B_{1994} , or there is >50% chance of this occurring within the next 5 years, if the TACC remains unchanged. NSW also uses model estimates of legal-sized biomass (B_L) under the same set of triggers.

Egg production – Estimated levels of retained egg production, determined from egg-per-recruit models, constitute a PI for assessing fishery performance against biological objectives in Tasmania, SA and WA. In Tasmania, no reference or trigger points are described for this PI, and this measure has not been calculated in recent assessments (Tarbath *et al.* 2002, Tarbath & Gardner 2009). The trigger for this PI in SA occurs when PREP is <50% of that estimated in the absence of fishing with the PI estimated from data available for those fishing areas from which a minimum percentage of the TACC has been harvested. In WA, the same trigger for PREP as in SA applies, but it is estimated for larger spatial units (Areas 2 and 3; Hart *et al.* 2010).

14.1.3.6 *Indicators obtained from other data sources*

PIs based on information obtained from sources other than fishery-dependent and fishery-independent data or model outputs are also used to assess fishery performance against biological objectives (**Table 4**). Most of these PIs apply only to SA, and include an annual assessment of stock status by divers (trigger is non-delivery of a report from industry), levels of illegal fishing (trigger is non-delivery of a report from compliance), number of prosecutions for illegal fishing (triggers are significant differences over the last two and five years) and the magnitude of the recreational catch (triggers are non-delivery of a report from PIRSA and a 25% change in recreational catch over a three year period).

SA, NSW and Victoria each have PIs for assessing fishery performance against environmentally-based biological objectives. PIs relating to disease (e.g. reported spread of disease and weight of catch discarded) are used in NSW and SA. The triggers include (1) the disease observed outside previous known locations and (2) $\geq 25\%$ increase in the weight of discarded catch. Compared with SA, Victoria and NSW have well developed ecological PIs. They are ‘indices of ecosystem health’ and ‘impact of fishing on the broader ecosystem’, respectively. SA and NSW also have PIs relating to the quality of the fishery-dependent data used to undertake assessment of the fishery against the biological objectives. In both states, these PIs are the percentage of the commercial fishery data received and subsequently entered into the catch and effort databases. The trigger points differ between states and are <100% and <90% of these data received and entered, respectively. SA also has a PI related to performance of the Research Agency, with that PI triggering if that agency fails to deliver a stock assessment report. Similarly, in NSW, there is a PI relating to adherence by the industry to their Code of Practice.

14.1.3.7 *PIs for assessment against economic objectives*

NSW, SA and Victoria have specified PIs for assessing fishery performance against the economic objectives of these fisheries, but there is little commonality among states (**Table 4**). Some PIs are based on fishery-dependent data, but most require data from a range of other sources including Government agencies, abalone processors and financial analysts.

14.1.3.8 *Indicators obtained from fishery-dependent data*

Catch, effort and CPUE are used to assess the economic performance of the abalone fisheries in SA, Victoria and NSW. In SA, a >25% increase in commercial fishing effort over a two-year period constitutes the trigger. For NSW, both catch and CPUE are used as economic indicators. For catch, the trigger is <85% of the TACC harvested, while for CPUE it is 85% of that observed in 1994 (the specified reference year). Victoria also uses CPUE as an economic indicator of fishery performance, but no trigger is specified.

14.1.3.9 *Indicators obtained from other data sources*

Both SA and NSW use PIs based on management costs to assess performance of the fishery against economic objectives. For SA, the trigger occurs when management fees exceed 10% of the gross value of production (GVP) or increase by >10% over two consecutive years. For NSW, the trigger is reached if fees increase by >CPI, again over two consecutive years. NSW uses beach price as an economic PI. The trigger occurs when the beach price falls below 85% of that in 2003/04, which was AUD35.70.kg⁻¹. Similarly, SA uses a five-year, negative trend in GVP as one of their economic PIs. This is closely related to the index used in Victoria, which is the gross profit of the fishery.

14.1.3.10 *PIs for assessment against governance objectives*

Only NSW and Victoria have PIs for assessing fishery performance against governance objectives (**Table 4**). In NSW, the governance PIs measure (1) delivery by service providers against contractual obligations, (2) provision of adequate data to inform the annual TACC setting process, and (3) currency of research, management and compliance plans. Victorian governance PIs measure total cost of management and degree of cost recovery. Both NSW and Victoria have compliance-related PIs for determining performance of the fishery against governance objectives. Those for Victoria (i.e. ‘compliance indices’) relate to both the commercial and illegal sectors, but are poorly described or defined. For NSW, the triggers are a >20% change in minor or >10% change in major offences by commercial fishers or those supporting the commercial catch (e.g. processors, exporters, etc.). Finally, Victoria has a PI measuring persistence and function of co-management arrangements.

14.1.3.11 *PIs for assessment against social objectives*

PIs to assess fishery performance against social objectives are described for the Victorian, SA and NSW abalone fisheries (**Table 4**). Several social PIs relate to the distribution of catch among sectors. For example, SA and NSW have PIs measuring the distribution of catch between the commercial and recreational sectors. The trigger for this distribution in NSW is a >25% change in the difference between the commercial and recreational catch over a five-year period. Similarly, in SA, the PI is triggered when the recreational catch increases by 25% over a three-year period. Allocation of indigenous catch is used as a PI in Victoria to recognise past access, but no trigger is prescribed for assessment. There are several other PIs prescribed for assessing fishery performance against social objectives. These include those for measuring the level of adherence by the commercial sector to their Industry Code of Conduct (SA and NSW), levels of compliance with fishery management regulations (SA), consultative participation among stakeholders in management decision making (SA, and NSW) and the community cost of resource use (Victoria).

14.1.3.12 *'Informal' PIs*

Current assessments of abalone fisheries use a range of data and analyses to help determine their status. In effect, these constitute 'informal' indicators because they provide additional measures by which fishery performance can be assessed against the management objectives of the fishery. As the primary focus of fishery stock assessments is biological, most of these indicators relate to assessing fishery performance against the biological objectives. Some of these 'informal' indicators have arisen from modification of 'formal' PIs over time. This has been required because of a lack of hierarchical structure (Chick *et al.* 2008, Mayfield *et al.* 2008a), poorly and arbitrarily assigned reference points (Tarbath *et al.* 2002, Chick *et al.* 2008, Upston *et al.* 2008, Tarbath & Gardner 2009) and changes to fishery management (e.g. TAC, size limits; Tarbath *et al.* 2002, Tarbath & Gardner 2009). In addition to these biologically-based indicators, a broad range of 'informal' indicators are used to assess economic performance in SA. In this section, we document and describe the biological and economic measures used to assess fishery performance that are not used as 'formal' PIs in Australian, state-based abalone fisheries (**Table 5**).

14.1.3.13 *Indicators obtained from fishery-dependent data*

Measures of fishery performance obtained from fishery-dependent data are commonly used in fishery assessment reports to provide context and aid assessments. At its simplest form, all states use current trends in catch, at various spatial scales, relative to long-term historical patterns, to help assess sustainability. For example, in SA, recent trends in catch are compared to historical average catches and historical patterns of fishing (e.g. cyclical fishing, historical maxima and minima; **Table 5**; Mayfield *et al.* 2008a, Chick *et al.* 2009). This approach is extended to CPUE and MDC, and suggests that PIs based on these analyses (e.g. current CPUE as a percentage of the contemporary maximum CPUE) may provide useful measures of stock status. Also in SA, temporal trends in the ratio of catch of different species, where there is no species-specific TACC (Western Zone, Region B), is used to provide a relative measure of species abundance. However, this index is likely to be less useful as a PI because other factors influence patterns of species composition in the commercial catch (e.g. differences in beach price).

Table 5. Summary of informal PIs used to assess management objectives in Australian abalone fisheries. Scale: Large (L) - entire fishery; Medium (M) - Zone/Region; Small (S) - smallest management unit.

Management Objective Category	Data source	Performance Indicator (general description)	Scale	State		
Biological	Fishery-dependent	Catch	Change in the distribution of catch relative to recent years and historical average	S, M, L	All States	
			Temporal trend in the catch ratio of different species	M	SA	
	Catch rate (CPUE)		Distribution of categorised CPUE (kg.hr ⁻¹) (skewness)	S, M	Tas	
			Mean monthly catch (MMC: kg.month ⁻¹) relative to Ref ₁₉₉₄	S	NSW	
	Weight of harvest		Weight of abalone harvested - Mean weight per Mth, Yr (Processor data)	M	NSW	
	Catch length structure		Length-frequency distribution - proportion 'large' and 'small'; change in shape (skewness)	S	SA	
	Shell morphology		Shape and appearance (algal fouling) of shells integrated to assess stock status	S	Vic	
			Length/ height ratio (above or below SL/SH = 3.25) - categorise stunted areas	S	SA	
			Doming length (to assess stock status and productivity)	S	Vic	
	Fishery-independent	Abundance/Biomass/Density		Absolute abundance & biomass (stratified sampling by CPUE); Harvest/risk decision table	S, M	SA
		Density		Density ratio - Legal:Sub-legal-sized	S, M	SA
		Length structure		Length-frequency distribution - persistence of length classes, distribution shape and correlation with CPUE	M, L	SA, WA, NSW
	Other data sources	Industry		Diver assessment of stock status	S, M, L	Tas, Vic, WA, NSW
				Decision tree of reef-scale stock status through rapid visual assessment (RVA) - Integrated fishery-dependent data with shell appearance	S	Vic
Economic	various		Exchange rate; Nominal beach price; Income, cost and profit per vessel; Gross operating surplus; Investment return; Economic rent	L	SA	

In Tasmania, recent assessments have included analyses based on temporal changes to CPUE distributions. This constitutes an alternative method of interpreting CPUE data, with changes in the statistical descriptions (e.g. skewness) of these distributions providing an index of stock status. NSW has also developed an alternative approach to using CPUE data, through estimation of a mean monthly catch (MMC; kg.month⁻¹), and consider this provides a more meaningful index of biomass than MDC or CPUE (Upston *et al.* 2008). MMC is assessed against that observed in 1994, the year fishery-independent surveys were initiated in the fishery.

NSW also uses the mean weight of abalone, by month and year, as an indicator of recruit abundance (**Table 5**; Upston *et al.* 2008). The NSW mean-weight measure is derived from a count of individual abalone in each 'bin' and total 'bin' weight. Changes in mean weight were interpreted with length-weight relationships to describe changes in the size structure of the commercially fished population, and emphasised the importance of recruits to the productivity of this fishery (Upston *et al.* 2008). Weight-grade data have allowed analogous measures to be developed for SA (see Section 2.2.4). SA abalone fishery assessment reports include values for the proportions of 'large' and 'small' legal-sized abalone in the commercial catch (**Table 5**; Mayfield *et al.* 2008a, Chick *et al.* 2009). In combination with the skewness of the distributions, these values are used to assess the level of reliance on new recruits to the fishery. Further development of these 'informal' indicators could lead to robust, 'formal' PIs in coming years.

A recent, novel approach to aiding reef-scale stock assessment in Victoria involves the combined use of shell morphology (doming and internal scarring), algal composition and cover on the dorsal surface of the shell and the length-structure of the commercial catch (Prince *et al.* 2008). The success of this approach suggests that these 'informal' indicators (i.e. shell morphology and algal cover) provide valuable information for assessing stock status, and could be considered as 'formal' PI in future years. Two additional indicators that should be considered, because of their apparent correlation with abalone age and maturity, are the ratio between shell length and shell height (Mayfield & Saunders 2008, Saunders *et al.* 2009a) and doming length (Day *et al.* 2010).

Indicators obtained from fishery-independent data

Three 'informal' indicators of stock status have been derived from fishery-independent data – all in SA (**Table 5**). First, FRDC project 2001/076 facilitated development of a survey method to estimate absolute, rather than relative, abalone abundance, from which estimates of harvestable biomass can be determined (McGarvey 2006, Mayfield *et al.* 2008b, McGarvey *et al.* 2008, Hart *et al.* 2009b, Mayfield *et al.* 2009). The primary outputs from these analyses are harvest decision tables used explicitly to set TACCs. The direct application of this method suggests survey estimated biomass would be a valuable 'formal' PI.

Second, the ratio between the density of legal-sized and sub-legal-sized abalone is used to provide a further indication of recruitment strength, and the potential capacity of the stock to support future catches. This ratio may also provide a useful 'formal' PI.

Third, the length-frequency data obtained on fishery-independent surveys provide an indication of recruitment strength and frequency, and are used to assess the resilience of populations to fishing. For example, the complete representation of all length classes in the

length-frequency distribution, below the MLL, provides strong evidence of regular recruitment, whereas the absence of some length classes is interpreted to represent reductions in recruitment. Similarly, relationships between pre-recruits to the fishery (i.e. all sub-legal-sized length classes within 15 mm SL of the MLL) and commercial CPUE the following year (Mayfield *et al.* 2008a) may also make a useful ‘formal’ PI.

14.1.3.14 *Indicators obtained from other data sources*

Over the last decade, abalone divers, licence holders and quota owners have increasingly contributed to management decisions, including amendment of size limits and TACCs. Their contributions have been underpinned by increasing (1) stewardship, (2) understanding of management objectives, and (3) awareness of data shortcomings and their implications for management. Whilst increased participation has occurred in all states, perhaps the most successful example is that of the Western Zone fishery in Victoria (Prince *et al.* 2008). Here, Government representatives and industry stakeholders collectively combine traditional fishery-dependent data (e.g. catch and effort) with measures of abalone population structure (based on shell morphology and shell fouling, see above) and perceived stock status to produce assessments of reef health. Following reef health assessment, a decision tree facilitates management decisions including changing size limits and amending TACCs. The importance of long-term diver knowledge of stock health has regularly been demonstrated during these discussions of stock status and harvest strategy (Prince *et al.* 2008). However, while the potential to include industry perceptions of stock status in a formal performance framework, with PIs and reference points, is established (Prince *et al.* 2008), the challenge is to convert this information into ‘formal’ PIs that can be objectively assessed.

SA commissions Econsearch Pty Ltd to undertake an annual economic analysis of the abalone fishery in SA. As part of their analyses, they use several indicators that are not used elsewhere as ‘formal’ PIs. These indicators include USD:AUD exchange rate, nominal beach price (i.e. corrected for CPI), cost per licence holder, income per vessel, costs per vessel, boat business profit, licence value, gross operating surplus, return on investment and economic rent. These indicators may make useful ‘formal’ PIs for assessing fishery performance against economic objectives in future years.

14.1.3.15 *‘Potential’ PIs*

Fishery assessment usually involves an ongoing process of continual improvement. Consequently, studies continue to explore more accurate, more precise and less costly approaches to assessing fishery performance, for the purposes of making management decisions. A recent suite of projects have focused on improving assessment for the purpose of management. These include FRDC 2001/074 – *Linking fishery-dependent and fishery-independent assessments of abalone fisheries*, 2004/019 – *Towards optimizing the spatial scale of abalone fishery assessment*, 2005/024 – *Abalone industry development: local assessment and management by industry*, 2006/029 – *Using GPS technology to improve data collection in abalone fisheries*, and FRDC 2007/066 – *Rapid response to abalone virus depletion in western Victoria: information acquisition and reefcode assessment models*. Other studies have been funded by state agencies. Collectively, these studies provide the opportunity to explore potential, novel PIs for abalone fisheries.

In this section, we document and describe potential PIs for assessing abalone fishery performance (**Table 6**). This information has been obtained from recent research reports and from discussions among managers, researchers and industry groups. As these indicators are less well developed than the ‘formal’ and ‘informal’ PIs considered above they are provided in less detail.

14.1.3.16 *Indicators that could be obtained from fishery-dependent data*

Spatial indices of stock status provide substantial potential for developing PIs from fishery-dependent data (Mundy 2010). The reduced size of GPS receivers, coupled with development of processes to capture, store and interrogate large volumes of data (i.e. millions of records per year) have allowed analyses of spatial data to be undertaken securely, efficiently and cost effectively (Mundy 2010). Linking the GPS stream to depth from a depth-temperature recorder (DTR) provides precise measures of effort, which are not captured by traditional logbook systems, and allows explicit identification of time periods when the diver is in the water. Separating GPS data between times when the diver is in the water or in the vessel is essential for reliably using the GPS data for any spatial analyses of fishing activity. It may also be possible to determine an index of searching effort.

Several ‘spatial’ indices are under consideration. Those based on Kernel-Density approaches for analysing and interpreting spatial data include Kernel Utilisation Distributions (i.e. spatial extent of fishing activity) and Kernel Density Indices (i.e. ratio of 50%:90% isopleths; Mundy 2010). Other potential indices are based on a grid-cell system. These include measures such as (1) number of cells fished, (2) divers per cell, (3) days fished per cell, and (4) the frequency distributions of fished cells (Mundy 2010). Detailed data describing diver effort with depth can differentiate types of fishing activity and separate dive events such as ‘search dives’ from ‘fishing dives’. The proportion of ‘search dives’ or the time allocated to searching may provide a valuable PI inferring changes in fishable stock through time. Further, more spatially-explicit catch reporting through time may also provide opportunities to obtain measures of fishery production such as yield per hectare and catch rates (e.g. $\text{kg}\cdot\text{m}^{-2}$).

Greenlip abalone in the Western Zone of the SA abalone fishery are routinely graded prior to the individual weight grades being weighed separately. These weight-grade data have been collected using a consistent method for >20 years, and because all catches are graded, these data are highly representative of the catch and the fishery (Mayfield 2010). Temporal changes in the composition of the grades provide meaningful measures of changes in the harvested stock (Mayfield 2010); consequently, weight-grade data are likely to be an informative PI of stock status.

Table 6. Summary of potential new performance indicators used to assess management objectives in Australian abalone fisheries. Scale: Large (L) - entire fishery; Medium (M) - Zone/Region; Small (S) - smallest management unit.

Data source	Performance Indicator (general description)	Scale
Fishery-dependent		
Catch & Effort	Catch & Effort - trends in spatial and temporal distribution (all scales - from integrated GPS and depth data)	S
Spatial indices of fishing activity	Kernel Density Indices - ratios of fishing area use 50%:90% isopleths	S
	Grid-cell Indices - number of cells; divers per cell; frequency distribution of fished cells	S
Weight of harvest	Mean graded meat weight (Processor data)	S, M, L
Fishery-independent		
Egg production	Egg production (derived from surveys of mature densities, biological data and population structure)	S, M
Survey density - mature	Temporal trend in clusters of mature stock	
- pre-recruits	Video surveys of density - pre-recruit abundance and catch relationship	S, M
- post-recruits	Proportional survival of new recruits >1 year - fishery reliance on recruitment	S, M
Model		
'Fishing to market' - specific lengths	Ratio of expected length classes - 'Fishing to market' of specific length classes to maximise economic yield	M, L
B_{MSY}	Maximum sustainable yield	M, L
B_{MEY}	Maximum economic yield	M, L
Other		
Industry	Diver assessment of stock health (obtained through a sociological survey)	S, M, L
Illegal fishing	Point estimate or trend in illegal catch	L
Ecological measures	Predator and competitor abundance	M, L
	Climate change - water temperature effects on distribution and biology	L
	Temporal change in area of habitat supporting populations	S, M

14.1.3.17 *Indicators that could be derived from fishery-independent data*

A direct measure of potential egg production is an informative index that could be determined directly from the fishery-independent surveys of abalone density and population structure. This approach is used in the SA prawn fisheries (Dixon *et al.* 2009), and relies on integrating the survey data with fecundity relationships. Similarly, the leaded-line survey method (McGarvey 2006, McGarvey *et al.* 2008) provides the opportunity to quantify clustering in abalone (McGarvey *et al.* 2010). Abalone are broadcast spawners, and males and females need to aggregate to maximise fertilisation success (Breen & Adkins 1980, Babcock & Keesing 1999). As increasing distances between individuals substantially reduces fertilisation rates (Levitan & Sewell 1998), quantification of clustering may be a particularly informative potential PI. Video-based, fishery-independent surveys (Hart *et al.* 2008) may also yield useful future PIs (e.g. providing data on pre-recruit abundance that can be used to predict future catches). Similarly, surveys of the survival of first-year recruits, through a dedicated tag-recapture program, could provide a PI to measure reliance of the fishery on recruitment.

14.1.3.18 *Indicators that could be derived from model outputs*

In addition to the use of reference points based on biomass and fishing mortality rates, existing length-structured, stock-assessment models could be modified to produce additional outputs that, in turn, could constitute valuable ‘formal’ PIs. For example, outputs of maximum sustainable yield (B_{MSY}) and maximum economic yield (B_{MEY}) have been proposed as potential PIs for the WA abalone fishery (Hart *et al.* 2009b). Further, if there were economic advantages to only fishing a particular range of length classes, perhaps for the live market, then a PI based on the expected ratio of these classes could be developed.

14.1.3.19 *Indicators that could be derived from other data sources*

Perhaps one of the greatest opportunities with re-considering PIs for abalone fisheries in Australia is the development of PIs based directly on industry knowledge and perception. This would provide one formal mechanism for incorporating ‘diver assessments of stock status’ into determining harvest strategies and harvest control rules. Whilst most fisheries currently use this information in conjunction with scientific assessments to determine TACCs, existing mechanisms tend to be ad hoc and informal. This approach may be most beneficial for areas of the fishery that do not support large catches and/or where information to assess fishery performance is more limited.

Similarly, although accurate estimates of illegal harvest are difficult to obtain (Gorfine *et al.* 2000), development of more informative levels of the magnitude of catch from this sector would provide a new, valuable PI for assessing stocks against biological objectives.

Whilst ecological impacts of harvesting abalone are likely to be small (Jenkins 2004), changes in habitat and the densities of predators and competitors could have more substantial effects (Andrew *et al.* 1998, Jenkins 2004, Barrett *et al.* 2009). For example, areas of reef inhabited by *Centrostephanus rodgersii* and converted to Barrens (*sensu* Underwood *et al.* 1991) no longer support substantial numbers of abalone (Andrew & Underwood 1992). Similarly, high densities of *Jasus edwardsii*, a known abalone predator, can

have a detrimental effect on the long-term abundance of blacklip abalone (Barrett *et al.* 2009). In future years, the threat of climate change, particularly rising water temperatures, may strongly influence abalone distribution and biology (Doubleday & Mayfield in prep). These patterns suggest that specific, ecologically-based PIs would be constructive as ‘formal’ PIs for future assessment of fishery performance against appropriate objectives.

14.1.4 Management objectives and PIs in abalone fisheries outside Australia

Haliotids are among the most intensively studied of exploited marine invertebrate species (Day & Shepherd 1995). Despite this extensive knowledge base, abalone fisheries outside Australia have proven difficult to manage sustainably (Prince 2004). Consequently, while numerous commercial fisheries became established across a broad geographical area spanning the USA (Davis *et al.* 1992, Tegner 2000, CDFG 2005), South Africa (Tarr 1992, DoEA 2009), Canada (Adkins 2000, Zhang *et al.* 2007), Sultanate of Oman (Johnson *et al.* 1992), Spain (Huchette & Clavier 2004), Gurnsey (Huchette & Clavier 2004), Japan (Leiva & Castilla 2001, Hamasaki & Kitada 2008), New Zealand (Hills 2009), Philippines (Tahil & Juinio-Menez 1999) and the Republic of Korea (Leiva & Castilla 2001), current commercial abalone fisheries, outside of Australia occur only in NZ (Hills 2009), Mexico (Guzman del Proo 1992, Ponce-Diaz *et al.* 2003) and France (Orstom 1992, Huchette & Clavier 2004). None of these abalone fisheries had or have management systems incorporating harvest strategies based on PIs with associated reference and trigger points. However, the Californian abalone fishery in the USA has an Abalone Recovery and Management Plan (CDFG 2005) that describes objectives and measures, which are similar to PIs with TRPs and LRPs and a draft harvest strategy has been developed for the NZ abalone fishery (Hills 2009).

In this section, we describe the biological management objectives and associated ‘formal and ‘informal’ PIs used for assessment of abalone fisheries outside Australia. Necessarily, this focuses on California and NZ, for which there are no social, economic or governance objectives or PIs. The information presented in this section was obtained from peer-reviewed, published literature, existing or draft fishery management plans or policy documents obtained directly from relevant fishery management agencies and stock assessment reports obtained from relevant research agencies.

14.1.4.1 USA – California

The Abalone Recovery and Management Plan (ARMP) describes objectives, PIs and reference points for achieving the short and long-term goals of the Californian abalone fishery (**Table 7**; CDFG 2005). The three objectives of the short-term goal are to (1) reverse the decline of populations, (2) rebuild populations to self-sustaining levels, and (3) re-establish self-sustaining populations capable of supporting a fishery. The PIs by which the first two objectives are assessed are based on survey measures of population size structure (90% of population between 100 and 177 mm SL or 25% of population >177 mm SL) and density (average: >0.2 abalone.m⁻² and >0.66 abalone.m⁻² in 75% of the survey area). Achievement of the size-structure (i.e. successful recruitment) and density (i.e. population maintenance and recovery) targets supports re-establishment of the fishery. Four PIs are prescribed for assessing performance against objective 3. These are maintenance of minimum population recruitment and density measures, CPUE and a serial depletion index.

This index, describes the average distance travelled between the access point and fishing ground. Each of these PIs has specified TRP & LRPs (**Table 7**).

The long-term goal, sustaining abalone populations and fisheries, has two key objectives. These are to (1) rebuild populations across the historical range of the fishery, and (2) establish long-term fishery management. The first objective is assessed by using PIs that measure the percentage of populations rebuilt to densities ≥ 0.66 abalone m^{-2} , and the percentage of areas closed to fishing (**Table 7**). PIs for assessment against the second objective are (1) developing fishing zones, (2) tagging each legally caught abalone and (3) ensuring adequate data for stock assessment (**Table 7**).

14.1.4.2 *New Zealand*

The NZ abalone (paua) fishery does not currently operate under a formal fishery management plan, but under a combined Ministry of Fisheries approved Harvest Strategy Standard (HSS) and the draft Paua Medium-term Research Plan (MTRP; Hills 2009). The HSS specifies setting of TACCs using model-based estimates of B_{msy} or conceptual proxies. The NZ paua fishery does not use B_{msy} . The two conceptual proxies, model-based estimates of spawning and recruited biomass (S_{ref} and B_{ref} , respectively), are described in the MTRP. A formal fishery management plan ('Fishplan') is currently being developed (Hills 2009). The potential management objectives and associated PIs of this 'Fishplan' are documented in the MTRP (**Table 7**). The two potential management objectives are (1) that productive capacity of stocks is sustained into the future at low levels of risk, and (2) maintenance of a healthy environment (**Table 7**; Hills 2009). The PIs for assessment of fishery performance against the first objective will potentially be based on model outputs (i.e. spawning biomass), ratio between catch and spawning biomass, and trends in catch, effort and CPUE. The general structure of 'soft' and 'hard' reference points (i.e. LRP and TRP) is outlined (see **Table 7**), but require further refinement.

PIs for assessing fishery performance against the environmental management objective are also being developed. These are related to measuring new pest/disease incursions, pest/disease spread and the biological diversity of paua sub-populations.

Table 7. Management objectives and PIs used in the Californian and New Zealand abalone fisheries.

Country	Management Objective	Performance Indicator	Target	Reference Points Trigger or Limit (H - Hard; S - Soft) (Management decision rule)
USA - California	<i>Short-term - Recovery and establishment of fishery</i> <i>Reverse decline of populations</i> <i>Rebuild populations to self-sustaining levels</i>	Population size distribution	90% population within intermediate range (100 - 177 mm) 25% of population in large range (>177 mm)	
		Population density	Minimum Viable Population (MVP) density >0.2 m ⁻² 75% of areas with density ≥0.66 m ⁻²	
		Population recruitment (Criteria 1)	Population recruitment of 100-177 mm range ≥0.45 m ⁻²	Density in all areas; ≥0.66 m ⁻² & deep areas >0.33 m ⁻² (H) (Re-establish fishing)
	<i>Reach sustainable fishery levels</i>	Population density (Criteria 2)	Population density in open Zones; >0.41 m ⁻² (deep) and >0.83 m ⁻² all areas Population density in closed Zones; >0.33 m ⁻² (deep) and >0.66 m ⁻² all areas	Density in all areas; >0.83 m ⁻² & deep areas >0.41 m ⁻² (H) (25% increase TAC) Density in all areas; ≤0.5 m ⁻² + No recruit. (H) (25% TAC reduction); ≤0.3 m ⁻² (H) (Area closure) Density in deep areas <0.25 m ⁻² + No recruit. (H) (25% TAC reduction)
		CPUE (Criteria 3)		Density in all areas; ≤0.25 m ⁻² (H) (TAC / Fishery closure) Statistically significant decrease in average CPUE compared with reference period (S)
		Serial depletion index (Criteria 3)		Increase in distance travelled (4-6 year average from 1990-1994) (S)
	<i>Long term - Fishery</i> <i>Rebuild populations across historic range</i> <i>Long-term fishery management</i>	Rebuild populations	≥75% of historical populations rebuilt to fishing density (≥0.66 m ⁻²)	≥50% areas closed (under density Criteria 2 or loss of habitat (H)) (Fishery closure)
		Zonal fishery boundaries		
		Distribute catch tags (individual tag for each legally caught abalone)		
		Data sources - Fishery-independent survey sites with Zones	Increase fishery-independent survey sites (representative of Zone)	
New Zealand	<i>Productive capacity of the stock sustained in to the future at low risk</i>	Current spawning biomass (S) relative to that in reference period (S/S _{ref})	S within range of S _{ref}	Probability that S is below a fixed % of S _{ref} is <10% or <5% (S) Probability that S is below a greater fixed % of S _{ref} is <2% (H)
		Current catch levels allow S to fluctuate around the reference period	Catch allowing S within range of S _{ref}	
		Trends in Catch & CPUE	100% TACC harvested CPUE maintained within range of CPUE _{ref} or Total hours consistent	Annual catch shows consistent decline over specified time (S) TACC can not be caught (H) CPUE shows consistent decline over specified time (S) Total hours shows significant increase over time (S)
	<i>Maintain a healthy environment</i>	Attempt to avoid pest/disease incursions	No new pest or disease incursions	
		Ensure disease/pest incursions are effectively managed	No spread of pests or diseases	
		Ensure biological integrity of wild stocks is not adversely affected by reseedling	Biological diversity of paua subpopulations is maintained	

14.1.4.3 *Other countries*

Information from abalone fisheries (extant or otherwise) outside Australia (section 2.2), California (section 2.3.1) and NZ (section 2.3.2) was difficult to obtain and invariably lacked management objectives, PIs and harvest strategies. In nearly all cases, this was because of the absence of formal fishery management plans. The difficulty of obtaining relevant management information was exacerbated for those abalone fisheries in Mexico, Japan and Chile (for the 'loco'). This is because these countries rely on national guidelines to direct the management of local fishing areas by local fishing co-operatives, each of which is responsible for area-specific management plans. These management plans are not readily available.

France (Huchette & Clavier 2004) and South Africa have initiated steps to develop harvest strategy models, similar to those in operation in Australia. In South Africa, the 'Abalone Scientific Working Group' recently proposed biological management objectives and associated PIs for this closed fishery. The management objectives are to (1) maintain or recover spawning biomass, and (2) reduce illegal fishing. The sole PI for the first objective is spawning biomass, for which the TRP and LRP are 40% and 20% of B_0 , over 15 years, respectively. For the second objective, the TRP is to reduce illegal fishing by a rate of 15% p.a..

Mexican Official Standards (Normas Oficiales Mexicanas, NOMs) outline national regulatory management tools (i.e. MLL, quota limits, gear specifications, seasonal and area closures and effort limitation) for management of local fisheries (Hernandez & Kempton 2003). Whilst it was not clear how management objectives and PIs were implemented, it appears assessments have included model-based estimates for the targeted extraction of ~25% of legal biomass (Ramade-Villanueva *et al.* 1998).

Similarly, in Chile, local co-operatives established within allocated Management and Exploitation Areas for Benthic Resources (MEABRs; Gallardo-Fernández 2008) have the responsibility to develop and implement management plans that include annual stock assessments of principal species (Gallardo-Fernández 2008). It is unknown if assessments include evaluation of fishery performance against 'formal' or 'in-formal' PIs.

14.1.5 Management objectives and PIs in other dive fisheries

Commercial dive fisheries exist for a large number of marine invertebrates, other than abalone. These range in scale and social and economic importance, from large export fisheries producing $>100 \text{ t.yr}^{-1}$ (e.g. sea cucumbers (Toral-Granda *et al.* 2008), sea urchins (Andrew *et al.* 2002) and queen conch (Theile 2001)), to small ($<10 \text{ t}$) developmental fisheries such as that for the Argentinean whelk (Narvarte 2006). Other fisheries include those for *Trochus spp.* (top shell), throughout Australasia (Stutterd & Williams 2003) and the Pacific Islands (Heslinga *et al.* 1984), the spiny lobster (*Panulirus argus*) fishery in Mexico (Bello *et al.* 2005) and throughout the Caribbean (Anon 1992), as well as many additional artisanal fisheries throughout the island countries of the Pacific, Indian and Atlantic Oceans.

There are strong similarities between abalone fisheries and other dive fisheries where the target species exhibit similar population structures. Consequently, we sought information on these fisheries to ensure that all opportunities to identify appropriate PIs were taken.

However, as with abalone fisheries outside Australia, relevant information on management processes for dive fisheries for other species (e.g. sea cucumbers, sea urchins, rock lobsters) was difficult to obtain and, again, much of the available documentation lacked clear management objectives, stipulated PIs and associated harvest strategies – despite formal fishery management plans for these fisheries being more common. We restrict our review to four fisheries – North American sea urchin fisheries, the Canadian sea cucumber fishery, the Western Australian pearl oyster fishery and the Queensland bêche-de-mer fishery – for which relevant information was available.

14.1.5.1 *North American sea urchin fisheries*

Few formal, sea-urchin management plans describe management objectives that are assessed using prescribed PIs and reference points (Anon 2009b, 2009c). Perry *et al.* (2002) describe the use of logistic-based models and the calculation of fishing mortality (F) at MSY as a LRP and that at 0.25-0.5 MSY as the TRP for the green urchin fishery. In the red urchin fishery, a survey-based approach is used to determine TACCs. Estimates of biomass obtained from fishery-independent surveys are multiplied by a conservative scaling factor (0.2), natural mortality (0.1), and a spatial scaling factor (i.e. fished bed area (Campbell *et al.* (2001) in Botsford *et al.* 2004). Fished-bed area is determined from detailed, fisher-based records of daily locations and catch available from a GIS database. For this fishery, Botsford *et al.* (2004) also propose use of lifetime egg production (LEP) as an approach to develop biological reference points.

14.1.5.2 *Canadian sea cucumber fishery*

Whilst the Canadian Sea Cucumber Fishery Management Plan contains a high number of management objectives that pertain to (1) the collection of biological information, (2) resource access among commercial, First Nation and recreational sectors, and (3) aquaculture and other policy development (Anon 2009a), no mechanism is described by which fishery performance is assessed against these objectives. Outcomes from a large-scale, adaptive management project (Hand *et al.* 2008) are helping to inform establishment of PIs, associated reference points and a sustainable harvest strategy. These PIs include sustainable harvest rates derived from a latent productivity population model (TRP: <6.7% of B_0 ; LRP: 50% B_0) within commercially viable fishing areas. This TRP aims to maintain the total population between 60 and 80% of B_0 , with the LRP triggering at 50% of B_0 .

14.1.5.3 *Western Australian pearl oyster fishery*

Annual TACCs in the Western Australian Pearl Oyster fishery are derived by assessing current catch rate with the mean catch rate of the ten-year reference period (1983-1992;(Hart & Murphy 2006). As changes in fishing efficiency now make this reference period redundant, new PIs and reference points have been developed (Hart & Murphy 2006). The three proposed PIs are (1) a recruitment index (0+ age class), (2) a settlement index (1+ age class), and (3) an abundance index. Current values of these PIs are evaluated against a 1996-2004 reference period. Collectively, these comprise a 'Forecasting Management Rule' that leads to clearly-defined TACC decision rules (Hart & Murphy 2006).

14.1.5.4 *Queensland bêche-de-mer fishery*

The Queensland bêche-de-mer Fishery operates under a fishery 'Performance Measurement System' which identifies management objectives, PIs, performance measures (triggers) and management responses (Anon 2008b). Management objectives encompass (1) biological (To ensure that fished stocks of sea cucumber are maintained at sustainable levels), (2) ecological, and (3) governance components of the fishery. Five PIs are used to assess fishery performance against the biological objective. These are catch, effort, survey completion, total biomass and legal biomass. The fishery-dependent PIs are assessed against spatially-specific, fishing-area targets. Failure to complete a triennial survey, a 15% reduction in total biomass and a legal biomass <50% of virgin are the triggers for the three PIs based on fishery-independent data. Management actions described in response to the triggering of PIs are broad, ranging from a review of the available data and re-assessment of the PIs, to fishing area closures and species specific reductions in TACC.

14.1.6 Discussion

Australian abalone fisheries share several commonalities. For example, they all began in the early to mid-1960's, each is managed with input and output controls – most notably limited entry, MLLs, ITQs and TACCs, and the principal export markets are China, Japan and Taiwan. However, despite these similarities, each state fishery developed and evolved independently, with these differences driven by variable state legislations and state-specific requirements. Consequently, there are also large differences among states in the way that these abalone fisheries are assessed and managed.

One of the key differences among states lies in the management objectives and PIs for assessment, and subsequent management, of the abalone resources. Whilst each state typically identifies biological, social, economic and governance objectives, their specificity, diversity and number vary considerably. The importance of ensuring the stocks are fished sustainably, coupled with the high product, licence and quota values, is reflected in the prescription of biological and economic management objectives for all Australian abalone fisheries. Only some states prescribe social (Tasmania, Victoria, NSW and SA) and/or governance (Victoria, WA and NSW) objectives.

Equivalent drivers (i.e. the same national and similar state-based requirements for biological and ecological sustainability) have provided the highest degree of consistency in the biological objectives. Thus, each state seeks to maintain or improve stock status, and to control impacts of fishing on the broader ecosystem. Nevertheless, differences are again apparent. For example, while Tasmania and Victoria have 'sustained productive capacity' as a biological objective, no mechanism to achieve this is specified. Biological objectives in SA, NSW and WA have more specific mechanisms, such as maintaining breeding stocks, spawning biomass and/or egg production. Ecologically-based, biological objectives also vary in focus among states – but likely aim to achieve the same outcomes. Objectives in Tasmania and Victoria focus on maintaining ecosystem health and ecological processes, whilst the focus in SA is on minimising the environmental impacts of fishing.

Economic objectives focus on ensuring economic development, economic viability and return on investments. Although implied in these objectives, SA and Victoria explicitly target cost reduction and economic efficiency as key management objectives. Governance and social objectives are much less similar among states than the biological and economic objectives. There is also some overlap among the economic, social and governance objectives. For example, WA and NSW have ‘cost-effective and efficient’ management practices as a governance objective, rather than as an economic objective as is the case in Victoria and SA. Similarly, reducing illegal fishing is identified as a social objective in SA and Victoria, but as a governance objective in NSW. These differences highlight the difficulty of clearly identifying and describing relevant economic, social and governance objectives for these fisheries. Resolving this difficulty would be advantageous because fisheries management is a complex process, requiring well-developed objectives to provide guidance for ensuring management decisions reflect the legislative requirements of ESD. Currently, few objectives are specific, with most being general in nature. The possibility of broad interpretations from these objectives makes determining their specific intents difficult. Management decision making would likely benefit from a re-evaluation, and regular consideration, of specific objectives for each fishery.

Selecting appropriate, robust PIs, which accurately measure changes in fishery performance, and are thus suitable for assessing abalone fisheries against the specified biological, economic, social and governance objectives is challenging. Nevertheless, in response to changes in biodiversity conservation legislation, recent management plans have PIs as a key feature. To date, there have been only low levels of agreement among stakeholders regarding the acceptance of PIs for abalone fisheries. Consequently, numerous PIs are used in the management of abalone fisheries in Australia (Gorfine *et al.* 2001). These differences occur despite similarities among many fisheries, particularly those for the same species. Whilst the lack of conformity reflects the difficulty of identifying appropriate PIs, the operational and legislative differences among fisheries, that reflect variable fishing histories and management arrangements among states, further exacerbate the lack of consensus. These differences mean use of the same PI set for each fishery is not necessarily appropriate, possible, or even desirable. It is also likely that different suites of PIs will be needed for different fishing areas, to most appropriately capture the high degree of spatial variation exhibited by haliotids (Prince *et al.* 2008, Mayfield & Saunders 2008).

As expected, most of the ‘formal’ PIs prescribed for Australian abalone fisheries are relevant to assessing fishery performance against biological objectives and, almost exclusively, against those objectives relating to sustainability rather than ecosystem integrity. Information for assessment is drawn from fishery-dependent and fishery-independent data and outputs from numerical models. As TACCs are the primary output control used to manage these fisheries, PIs based on catch are commonly used to assess fishery performance against biological objectives. Approaches include change in total catch, catch as a percent of TACC, catch in relation to fishing history and change in the spatial distribution of catch. Catch rates are also commonly used as PIs. Two of these derived, and inferred relative measures of stock abundance are typically employed: mean daily catch and mean catch per hour (CPUE). Model-derived (stock assessment and/or egg production) PIs are also used in each state. SA, WA and Tasmania also use measures of the length structure of the commercial catch as biological PIs. In contrast, the limited use of fishing effort suggests it is unlikely to be an informative PI. Similarly, PIs based on fishing mortality and standardised CPUE are rarely used to assess fishery performance. Fishery-independent, survey measures of abalone abundance are also less common, only being prescribed as PI

in SA and WA. Whilst all states have ecologically-based, biological objectives, only SA, NSW and Victoria have prescribed PIs for assessing fishery performance against these. In most cases, there are few data for assessing fishery performance against these ecological PIs and objectives (Anon 2007b).

Specified PIs for assessing fishery performance against economic, social and governance objectives are generally less well developed than those for assessment against the biological objectives. There is also often overlap among states with these objectives and PIs. Catch, effort and CPUE are used as PIs to assess economic performance of the abalone fisheries in SA, Victoria and NSW. SA and NSW also have PIs based on management costs to assess performance of the fishery against economic objectives. These include management fees as a percentage of GVP and the rate of fee increases.

Only NSW and Victoria have PIs for assessing fishery performance against governance objectives. These governance PIs measure levels of service delivery, level of data to inform the annual TACC setting process, currency of research, management and compliance plans, total cost of management and degree of cost recovery. PIs to assess fishery performance against social objectives are described for the Victorian, SA and NSW abalone fisheries. Most relate to the distribution of catch among sectors, adherence by the commercial sector to their Industry Code of Conduct, and level of stakeholder consultation in management decision making. Social objectives and associated PIs for fisheries are likely to be substantially improved by a current FRDC-funded project (2010/040).

The suitability of this broad range of 'formal' PIs used in the management of Australian abalone fisheries to act either as an 'early warning signal' or as an indicator of improving resource status is poorly understood. As management decisions for these fisheries are collectively informed by the PIs, future sustainability of these fisheries requires that these PIs be informative. Thus, these PIs must provide clear, timely indications of variation in abalone abundance and/or population structure, and be sensitive to, and effective at, detecting biologically-meaningful changes. In the absence of this level of sensitivity, these PIs will fail to identify Zones/Regions/Reefs where the resource could sustain additional fishing pressure, or may be overfished.

Uncertainty in the quality of the PIs has limited development of formal, harvest-control rules that prescribe clearly-defined management outcomes. Thus, there is no formal link between PI-based assessments of stock status and management responses. Consequently, harvest strategies for these fisheries are not fully developed, and it is frequently difficult for stakeholders to agree on appropriate management responses (TACC increases or decreases), including their magnitude and timely implementation (Caddy 1998).

Concerns over PI quality have also resulted in assessments of stock status being only loosely based around the prescribed PIs. For example, in SA, the prescribed biological PIs form only one component of the broad range of information used by the research agency in determining stock status. This approach has evolved because SA has a large number of biological PIs that have frequently triggered in opposing directions, thereby providing contrasting inferences regarding stock status. In several cases, analyses to assess fishery performance against prescribed biological PIs have been discontinued. For example, length-structured, stock-assessment models are no longer used in Victoria to evaluate the relationship between current estimates of mature biomass (B_m) and those for the prescribed reference period. In Tasmania, PIs describing levels of egg production have not been used

in the assessment of biological objectives because Tarbath *et al* (2002) conclude "...it is unlikely that maintaining levels of egg production alone would ensure a sustainable fishery".

Deviations from the prescribed assessments of fishery performance using the mandated PIs, have led to the development and use of 'informal' PIs. This move has been required because the need for assessing stock status has not diminished, despite the absence of a reliable and accepted set of PIs. As stock assessment reports are predominately biologically focused, most of these 'informal' PIs are suited to assessing fishery performance against the biological objectives. 'Informal' PIs have been developed in all states. The most common of these is evaluating changes in the distribution of catches relative to recent years and historical averages. Several 'informal' PIs are extensions or adaptations of 'formal' PIs used elsewhere. For example, Tasmania and NSW use frequency distributions of CPUE and mean monthly catch, respectively, to help inform stock status. Similarly, SA uses proportions of small and large abalone in the catch, as an additional measure above that prescribed (i.e. mean length). The most novel, 'informal' PIs are those that are currently being derived from the shells of commercially-harvested abalone in Victoria (Prince *et al.* 2008, Day *et al.* 2010). Here, shell morphology (i.e. internal scarring and doming) and the algal composition and cover on the shells' dorsal surface are used as an index of fishing pressure and retained egg production. In combination with the length-structure of the commercial catch, this approach has been employed in determining spatially-specific voluntary size and catch limits. Suitable PIs from these data may include doming length (Day *et al.* 2010) and the length-height ratio (Mayfield & Saunders 2008, Saunders *et al.* 2009a). Estimates of biomass from fishery-independent surveys that measure absolute abundance (McGarvey *et al.* 2008) are also used as 'informal' PIs to aid stock assessment (Mayfield *et al.* 2008b) and determine TACCs.

Pursuit of continued improvement and more cost-effective approaches to undertaking abalone assessments has led to the exploration of additional, potentially suitable PIs for these fisheries – with much of this work supported by a group of recent, FRDC-funded projects (2004/019, 2005/024, 2006/029 and 2007/066). Perhaps the most suitable potential PIs will be obtained from spatial analyses underpinned by the combination of GPS and depth-temperature-recorder data (Mundy 2010). Aside from developing a PI reflecting 'searching' effort, this system allows exploration and evaluation of Kernel-Density- (e.g. ratio of 50%:90% isopleths) and grid-cell-based PIs (e.g. number of cells fished, divers per cell) for assessing stock. Other potential PIs that have recently been explored include a measure of abalone clustering (McGarvey *et al.* 2010), direct measures of egg production from fishery-independent surveys (Dixon *et al.* 2009) and video-based, pre-recruit abundance surveys (Hart *et al.* 2008). Development of PIs based directly on industry knowledge and perception would provide a mechanism for incorporating this information into harvest strategies and the development of harvest control rules. Formalising incorporation of this information into any TACC-setting process would reduce their current ad hoc and informal nature. Nevertheless, whilst this approach provides a strong opportunity for advancement, the mechanism by which it can be achieved is not readily apparent.

Retention of 'formal' PIs in Management Plans, legislation and regulation, in concert with the development of 'informal' PIs and the ongoing evolution of new PIs, has resulted in a large and diverse array of potentially informative PIs for Australian abalone fisheries (**Table 4 – Table 6**). As a review of management objectives and associated PIs in abalone fisheries outside Australia, and in other dive fisheries, did not yield any additional PIs, the

‘formal’, ‘informal’ and potential PIs employed in assessing Australian abalone fisheries were amalgamated into a comprehensive ‘working list’ (**Table 8**). This list is structured by PI category (i.e. biological or economic) and then by data source. The focus is on biological and economic PIs, as the social and governance PIs were less well defined and, by necessity, some PIs are not well described. A total of 56 PIs have been identified.

The next challenge will be to determine those PIs that are likely to be most informative, and consequently most suited, for use in assessment of Australian abalone fisheries. This process will be undertaken through a number of steps. First, expert panels in SA and Tasmania will evaluate this ‘working list’. These results will be collated and merged with comments solicited from researchers, managers and industry representatives in WA, NSW and Victoria (Section 3). This process will conclude with a more restricted list of PIs that comprise the most valued amongst stakeholders. This restricted list of PIs will be evaluated against known fishery performance (step 2; Section 4) and using a management strategy evaluation (step 3; Appendix 4). In combination, these steps will provide an analysis of the value of these PIs and management strategies (i.e. management responses) that will direct their adoption for informing ongoing sustainability of Australian abalone fisheries.

Subsequent to these steps, other issues relating to the PIs will need to be considered. This may best be achieved separately by each state, because operational and legislative differences among fisheries mean that PI use is unlikely to be uniform. The primary issues requiring consideration are (1) number and diversity of PIs required, (2) spatial scales of application, (3) using appropriate data, (4) analytical approach, (5) integration of PIs into a single ‘index’ of stock status, and (6) development of prescriptive harvest control rules.

A good example of a Management Plan with a large number of PIs is that for the SA abalone fishery. The most recent assessment of the Western Zone required assessment against 106 and 92 PIs for blacklip and greenlip, respectively (Chick *et al.* 2009). While the diverse suite of spatially-structured PIs affords two key advantages – almost all aspects of the fishery are encompassed and the spatial focus reflects complex abalone population structures (Saunders & Mayfield 2008, Miller *et al.* 2009, Saunders *et al.* 2009a, Saunders *et al.* 2009b) – assessments are complicated because those PIs that trigger seldom provide consistent inferences about stock status. These problems could be overcome by selecting the most informative PIs and arranging them hierarchically, based on the ‘traffic-light’ or ‘thermostat’ approaches suggested by Caddy (2002) and Shepherd & Rodda (2001).

There is growing acceptance of the need to match the scale of management with that of biologically functional units. This change is required because recent studies have confirmed that independent, abalone populations exist at much smaller spatial scales than those over which many of the fisheries are currently managed (e.g. Prince 2004, Morgan & Shepherd 2006, Mayfield & Saunders 2008, Miller *et al.* 2009). Thus, many key fishing areas constitute groups of large areas of reef, separated by kilometres, that are likely to function as semi-independent biological populations (meta-populations), with each supporting high catches. Good examples of this are fishing area 9 in SA (~120 t.yr⁻¹) that comprises three distinct fishing grounds (Ward Island, Hotspot and Flinders Island), and block 5 in north-western Tasmania (~100 t.yr⁻¹) that may contain up to 30 separate, productive reefs (discrete headlands, cliff lines, and offshore reefs). This change will require PIs to be applied and appropriate reference and trigger points defined, at scales previously not considered. One complication with this approach, that will need to be overcome, is the limited resources for data collection, reporting and management at these scales. Different

approaches are likely to be required in different fisheries. Thus, whilst it is likely that the Western Zone in Victoria will continue to pursue reef-scale assessment and management, the scale of the Tasmanian fishery may require selection of fine-scale assessment with a larger scale of management. In part, this challenge could be overcome by tailoring assessment requirements to the importance of each fishing unit.

Table 8. Summary of formal, informal and potential PIs identified in the review of Australian abalone fisheries, abalone fisheries elsewhere and other dive fisheries. The use indicates the number of States using this approach (max = 5), with ‘new’ identifying those not currently employed.

Management Objective	Data class	Datatype	Performance Indicator	Use (rank 1-5)	
Biological	Fishery-dependent	Catch	Catch as % of TACC	4	
			Catch as % of expected catch - by fishing area	2	
			Temporal patterns in catch	5	
			Change in spatial distribution of catch	2	
			Catch ratio between species	1	
			Mean individual weight	1	
			Proportions of catch by grade	New	
		Effort	Diver days	1	
			Total hours	1	
			hr day ⁻¹	1	
		Catch rate	hr ⁻¹ - raw	5	
			hr ⁻¹ - standardised	1	
			hr day ⁻¹	2	
			hr month ⁻¹	1	
	Spatial	Grid-cell indices - no. of cells fished, no. divers, cell ⁻¹ , etc	New		
		Kernel Density Index - ratio of 50%/90% isopleths, etc	New		
	Catch length structure	Mean or Median length	3		
		Proportion of 'large' and 'small' length classes	1		
		Change in shape (sheariness)	1		
		Shell shape & appearance - dom. in: length: internal scarring: algal fouling: length:height:ratio	1		
		Ev ⁻¹	1		
		Fishery-independent	Density	Density of length classes - total, legal, sub-legal, mature, immature, recruits, pre-recruits, etc	4
				Legal/sub-legal-sized density ratio	1
	Proportion of pre-recruits surviving in first year			New	
	Patch size and density of mature stock			New	
	Biomass		Exploitable biomass	1	
			Length structure	1	
	Reproductive potential		Proportion of 'large' and 'small' length classes	1	
Change in shape (sheariness)			5		
Length class representation			1		
Total potential egg production			New		
Model outputs	Biomass (population model)	Biomass - mature, legal, exploitable, etc	3		
		Exploitation rate	New		
		Maximum economic yield	1		
	Egg production	Maximum sustainable yield	1		
		Percentage retained egg production	2		
Other	Industry	Diver assessments of stock status	5		
		Illegal catch	No. of prosecutions	1	
	Recreational catch	Catch as % of TACC	New		
		Catch as % of TACC	New		
		Density of predators and competitors	1		
	Ecological	Water temperature	New		
		% of fishing areas affected - demonstrable ecological change from fishing	1		
Economic	Fishery-dependent	Catch	% of TACC harvested	1	
			Effort	Total hours	1
			Catch rate	hr ⁻¹	1
	Other	Commercial business costs	hr day ⁻¹	New	
			hr month ⁻¹	New	
			Management costs - %GVP, %asset value, etc	3	
			Beach price	1	
			\$US-\$A exchange rate	1	
			Gross operating surplus	1	
			% Return on investment	1	
			Economic rent	1	
			Operating costs	1	
			Net profit	1	

The data and methods used to calculate PIs should be well considered and clearly defined. This should incorporate definitions for minimum data and sample size requirements. This approach will help to ensure that application of the PIs is relevant and that high levels of statistical power do not outweigh biologically relevant changes. It is also important to consider how the PI will relate to management responses. The inclusion of target and limit reference points would greatly aid in the interpretation of changes in data and may also provide clearer direction for subsequent management decisions (Sainsbury 2008). Eliminating PIs referring to inter-annual changes and retaining those determined over longer time periods may also help to reduce inconsistency in their interpretation, and avoid unnecessarily responding to ‘noise’ in the fishery.

Integrating the PIs into a single measure of stock status – by metapopulation, fishing area or zone – is likely to further clarify required management responses as a consequence of changes in the PI. Thereafter, developing prescriptive harvest control rules that describe management responses will also likely overcome the frequent difficulty of stakeholders agreeing on appropriate management decisions (TACC increases or decreases). For abalone in Australia, such a system is only currently operating for WA. There, sCPUE and fishing mortality are considered collectively in a risk-assessment matrix; high levels of risk lead to prescribed TACC reductions, whilst low levels of risk allow prescribed increases to the TACC (Hart *et al.* 2010). SA have been working towards this outcome, but have taken a different approach. The current proposal is to assess stocks within biologically relevant areas (Management units; MU) using PIs that are weighted (based on reliability and accuracy to inform stock status) and scored (based on results from short, medium and long-term triggers). Weighted scores for all available PIs would be summed to generate a probability of stocks within a MU being overfished. Management units will be categorized within the range of probabilities by total score, with each category having minimum prescriptive management actions based on this assessment of biological status.

14.2 Qualitative assessment of PI by expert panels

14.2.1 Introduction

The review of Management Objectives and Performance Indicators (PIs) for abalone fisheries in Australia (and dive fisheries elsewhere) undertaken in Section 2 yielded 56 potentially-informative PIs (see **Table 8**), termed a ‘working list’. As the social and governance management objectives, and associated PIs, were poorly defined and described, the 56 PIs identified relate primarily to biological and economic management objectives. These 56 PIs are currently mandated (i.e. ‘formal’), used to assess stock status (i.e. ‘informal’) or proposed (i.e. ‘potential’) and are dependent on a diverse array of data sources.

Clearly, use of such a large number of PIs for the management of Australian abalone stocks is untenable. Consequently, those PIs that are likely to be most informative, and therefore most suited, for use in assessment of these fisheries need to be determined. The first step in this process was the evaluation of this ‘working list’ at workshops in SA and Tasmania.

Subsequent to the evaluation of PIs at these workshops, a more restricted list of PIs, that comprise the most valued amongst stakeholders, will be evaluated against known fishery performance (Section 14.3) and using a management strategy evaluation (Appendix 4). In combination, these steps will provide an analysis of the value of these PIs and management strategies (i.e. management responses) that will direct their adoption for informing ongoing sustainability of Australian abalone fisheries.

The objectives of this Section are to (1) describe the expert panel review process; (2) document the findings; and (3) develop a short list of PIs deemed most informative.

14.2.2 Methods

Three workshops were convened (see details and lists of attendees below). Workshop participants consisted of individuals experienced in the application and interpretation of PIs at zone, state and regional scales and, hereafter, are referred to as an expert panel. Each

expert panel typically comprised all stakeholders – licence holders, divers, industry representatives, state fishery managers and researchers. The expert panels were numerically dominated by industry.

These expert-panel workshops followed a request for the same information made to NSW, Victoria and WA via email (19 May 2010). As no feedback was received from these three States, the remainder of this Section focuses exclusively on the information obtained at the workshops in SA and Tasmania.

At each workshop, participants were provided with a description of each PI, its use and interpretation, and then asked to rank it as very useful (VU), useful (U), some use (SU) and not useful (NU). Several PIs at each workshop were not ranked because they were either not applicable (NA) to that jurisdiction or, in the case of workshop 2, considered individual business tools that were irrelevant to the assessment of fishery performance.

To identify those PIs considered most informative across the three workshops, scores were assigned to each category (VU – 4; U – 3; SU – 2; NU – 1 and NA – 0). Thereafter, scores for each PI were summed and expressed as a percentage of the maximum possible score. As PIs that were ranked at only one workshop were excluded from this analysis, the maximum possible scores were 12 (i.e. no NA) or eight (one NA). Percentages of the maximum possible score for those PIs with maximum possible scores of eight are unavoidably positively biased. Workshop participants were also asked to identify any additional PIs that could also be beneficial for assessing abalone fisheries.

Workshop 1 – Hobart, Tasmania, 17 June 2010: M Bradshaw, I Cartwright, B Cobbing, C Gardiner, R Day, M Haddon, A Hansen, R Herweynen, J Huddleston, T Johnston, D Lison, P McKibben, R Rex, S. Mayfield, D Tarbath, N Wallace, P van der Woude and G Woodham.

Workshop 2 – Mt Gambier, SA, 20 July 2010: C Carrison, J Cope, J Godden, A Hogg, A Leopold, A Martel, S. Mayfield, R McEwan, C Noell, G Pollard, M Snart and B Tarr.

Workshop 3 – Pt Lincoln, SA, 6 August 2010: D Buckland, T Craig, J Dent, D Edmunds, B Ford, D Matthews, S. Mayfield, S Miller, B Stobart and J Woolford.

14.2.3 Results

Collectively, expert panel members identified an additional 18 potential PIs (**Table 9**). Most of these (10) related to assessing the economic performance of the fishery and included lease price, performance gap (i.e. difference between current net present value and optimal net present value) and market capitalisation. Four of the remaining eight PIs were for assessing illegal and recreational catch. Thus, a total of 75 PIs were considered at each of the three workshops.

Three PIs – raw CPUE ($\text{kg}\cdot\text{hr}^{-1}$), proportions of large and small length classes in the commercial catch and diver assessment of stock status – were ranked as VU at each of the three workshops and, therefore, attained a total score of 12 (**Table 9**). Two PIs relating to the distribution and structure of the commercial catch (i.e. temporal patterns in catch and mean or median length) had total scores of 11 as they were ranked VU at the Tasmanian and SA SZ workshops and U at the SA WZ workshop. A further five PIs had total scores of 10. These PIs included the percent of TACC harvested, total hours, the density of

length classes in fishery-independent surveys and two ecological PIs (i.e. water temperature and the percent of fishing areas affected by disease). For each of these five PIs their ranked categories were identical at SA workshops which, in turn, differed from the ranking assigned at the Tasmanian workshop. Four PIs were also ranked as NU by each expert panel. These PIs were catch ratios between species, CPUE ($\text{kg}\cdot\text{month}^{-1}$) $\text{F}\cdot\text{yr}^{-1}$ and the number of prosecutions as a measure of illegal catch.

Apart from the three PIs that scored a total of 12 (i.e. VU at each workshop) and those with a total score of 3 (i.e. NU at each workshop), there was generally little consistency in the outcomes from the three workshops. For example, a large proportion of PIs were either not ranked or were deemed not applicable by some expert panels. Obvious amongst these were all economic PIs at the SA SZ workshop and PIs relating to model outputs and spatial measures of fishery performance at the SA WZ workshop.

Participants in the Tasmanian workshop identified 41 PIs as VU, whilst at the two SA workshops the number of PIs identified as VU were considerably smaller (eight and 15). Several PIs were also ranked differently at each of the three workshops. For example, two PIs – shell shape and appearance and difference between mean/median length and size-at-maturity – were ranked VU at the first SA workshop, U at the Tasmanian workshop and NU at the second SA workshop. The reverse also occurred: change in shape of the commercial length structure, each of the four PIs related to fishery-independent measures of population length structure and egg production were ranked NU at the Tasmanian workshop and either VU or U at the two SA workshops.

There were also several state-based differences. Sixteen PIs were ranked the same at the two SA workshops, but differently at the Tasmanian workshop. The largest differences were for standardised CPUE ($\text{kg}\cdot\text{hr}^{-1}$), diver days, $\text{hr}\cdot\text{day}^{-1}$ and $\text{kg}\cdot\text{day}^{-1}$ which were ranked VU in Tasmania, but NU at both SA workshops.

The percentage of maximum score, determined by scoring the ranks provided to each PI at each workshop, provides an objective approach to synthesising the data. For the biological PIs, 23 were $\geq 75\%$ of the maximum score (**Figure 8**). Of these, there were three at 100%, two $>90\%$ and a further six at $>85\%$. The lowest value, 25%, was attributable to four PI. Whilst fewer economic PIs were ranked, one – GVP – had a value of 100% and a further eight were $>85\%$ (**Figure 9**).

Table 9. Summary of formal, informal and potential PIs ranked by three expert panels. ‘New – R’ and ‘New – EP’ indicate PIs not currently in use that were identified by the review (R; Section 14.1.2) or by the expert panels (EP), respectively. PIs ranked ‘very useful’, ‘useful’, ‘some use’ and ‘not useful’ are hierarchically scored 4 to 1 and shaded green, yellow, orange and red, respectively. PIs not ranked or deemed not applicable are shaded black.

Management Objective	Data class	Datatype	Reference	Performance Indicator	Use (rank 1-5)	EP1 - Tas	EP2 - SA&Z	EP3 - SA(WZ)	Score	Max. score	% Max. score
B i o l o g i c a l	Fishery-dependent	Catch	1	% of TACC harvested	4	2	4	4	10	12	83.3
			2	Catch as % of expected catch - by fishing area	2	4	4	1	9	12	75.0
			3	Temporal patterns in catch	5	4	4	3	11	12	91.7
			4	Change in spatial distribution of catch	2	2	3	3	8	12	66.7
			5	Catch ratio between species	1	1	1	1	3	12	25.0
			6	Mean individual weight	1	2	3	3	8	12	66.7
			7	Proportions of catch by grade	New - R	1	1	4	6	12	50.0
		Effort	8	Diver days	1	4	1	1	6	12	50.0
			9	No. of drops	New - EP	3	3	1	7	12	58.3
			10	Total hours	1	4	3	3	10	12	83.3
			11	Ht by depth	New - EP	3	1	1	5	12	41.7
			12	Ht day ⁻¹	1	4	1	1	6	12	50.0
		Catch rate	13	Kg hr ⁻¹ - raw	5	4	4	4	12	12	100.0
			14	Kg hr ⁻¹ - standardised	1	4	1	1	6	12	50.0
			15	Kg day ⁻¹	2	4	1	1	6	12	50.0
			16	Kg month ⁻¹	1	1	1	1	3	12	25.0
		Spatial	17	Grid-cell indices - no. of cells fished, no. divers cell ⁻¹ , etc	New - R	4	3	0	7	8	87.5
			18	Kernel Density index - ratio of 50%:50% isopleths, etc	New - R	4	3	0	7	8	87.5
			19	Swept area	New - EP	4	3	0	7	8	87.5
		Catch length structure	20	Mean or Median length	3	4	4	3	11	12	91.7
			21	Proportion of 'large' and 'small' length classes	1	4	4	4	12	12	100.0
			22	Change in shape (skewness)	1	1	3	3	7	12	58.3
			23	Shell shape & appearance - doming length, internal scarring, algal fouling, length/weight ratio	1	2	4	1	7	12	58.3
			24	Difference between mean/median length and size at maturity	New - EP	2	4	1	7	12	58.3
			25	F _{yr} ⁻¹	1	1	1	1	3	12	25.0
	Fishery-independent	Density	26	Density of length classes - total, legal, sub-legal, mature, immature, recruits, pre-recruits, etc	4	2	4	4	10	12	83.3
			27	Legal/sub-legal-sized density ratio	1	2	4	3	9	12	75.0
			28	Proportion of recruits surviving in first year	New - R	2	4	3	9	12	75.0
			29	Patch size and density of mature stock	New - R	2	4	1	7	12	58.3
		Biomass	30	Exploitable biomass	1	1	1	3	5	12	41.7
		Length structure	31	Proportion of 'large' and 'small' length classes	1	1	3	4	8	12	66.7
			32	Change in shape (skewness)	4	1	4	3	8	12	66.7
			33	Length class representation	1	1	4	3	8	12	66.7
		Reproductive potential	34	Total potential egg production	New - R	1	3	3	7	12	58.3

Table 9

continued.

Management Objective	Data class	Data type	Reference no.	Performance Indicator	Use (rank 1-5)	ER - Tar	EP2 - SA(SZ)	ER3 - SA(WZ)	Score	Max. score	% Max. score	
B i o l o g i c a l	Model output	Biomass (population model)	33	Biomass - mature, legal, exploitable, etc	3	4	2	0	6	8	750	
			34	Exploitation rate	New-E	4	3	0	7	8	875	
			37	Total potential egg production	New-E	4	3	0	7	8	875	
			38	Maximum economic yield	1	4	0	0	4	-	-	
			39	Maximum sustainable yield	1	4	3	0	7	8	875	
		Egg production		40	Percentage of total egg production	2	1	3	3	7	12	583
	Other	Industry		41	Linear assessment of total status	5	4	4	4	12	12	1000
				42	No. of prosecutions	1	1	1	1	3	12	250
		Illegal catch		43	Linear assessment	New-EP	4	3	2	9	12	750
				44	Police assessment	New-EP	3	0	0	3	-	-
				45	Compliance assessment	New-EP	0	3	3	6	8	750
				46	Catches % of IACC	New-E	3	3	0	6	8	750
		Recreational catch		47	No. Recreational fishing licenses	New-EP	3	0	0	3	-	-
				48	Catches % of IACC	New-E	3	3	1	7	12	583
		Ecological		49	Diversity of predators and competitors	1	3	3	1	7	12	583
				50	Water temperature	New-E	4	3	3	10	12	833
				51	% of fishing gear affected - demonstrate ecological change from fishing	1	4	1	1	6	12	500
			52	% of fishing gear affected - disease detection and spread	1	4	3	3	10	12	833	
E c o n o m i c	Fishery-dependent	Catch	53	% of IACC harvested	1	1	0	1	2	8	250	
			54	% Live vs % Canned	New-EP	4	0	0	4	-	-	
			55	% IACC/linear	New-EP	4	0	0	4	-	-	
			56	GV/P	New-EP	4	0	4	8	8	1000	
		Effort		57	Total cost	New-EP	4	0	3	7	8	875
				58	Total hours	1	1	0	1	2	8	250
		Catch rate		59	Kg/ki'	1	4	0	1	5	8	425
				60	Kg/dy'	New-E	4	0	1	5	8	425
			61	Kg/month'	New-E	1	0	1	2	8	250	
	Other	Commercial business cost		62	Management cost - %GV/P, %cost value, etc	3	4	0	3	7	8	875
				63	Royalties	New-EP	4	0	0	4	-	-
				64	Market capitalization	New-EP	4	0	3	7	8	875
				65	Performance gap (current net present value vs optimal net present value)	New-EP	4	0	3	7	8	875
				66	License price	New-EP	4	0	0	4	-	-
				67	Boat price	1	4	0	1	5	8	425
				68	US\$A exchange rate	1	4	0	3	7	8	875
				69	Costs operating expense	1	4	0	1	5	8	425
			70	% Return on investment	1	4	0	3	7	8	875	
			71	Economic rent	1	4	0	1	5	8	425	
			72	Operating costs	1	4	0	3	7	8	875	
			73	Net profit	1	4	0	3	7	8	875	
	74	Sale price	New-EP	0	0	3	3	-	-			
	75	License value	New-EP	0	0	3	3	-	-			

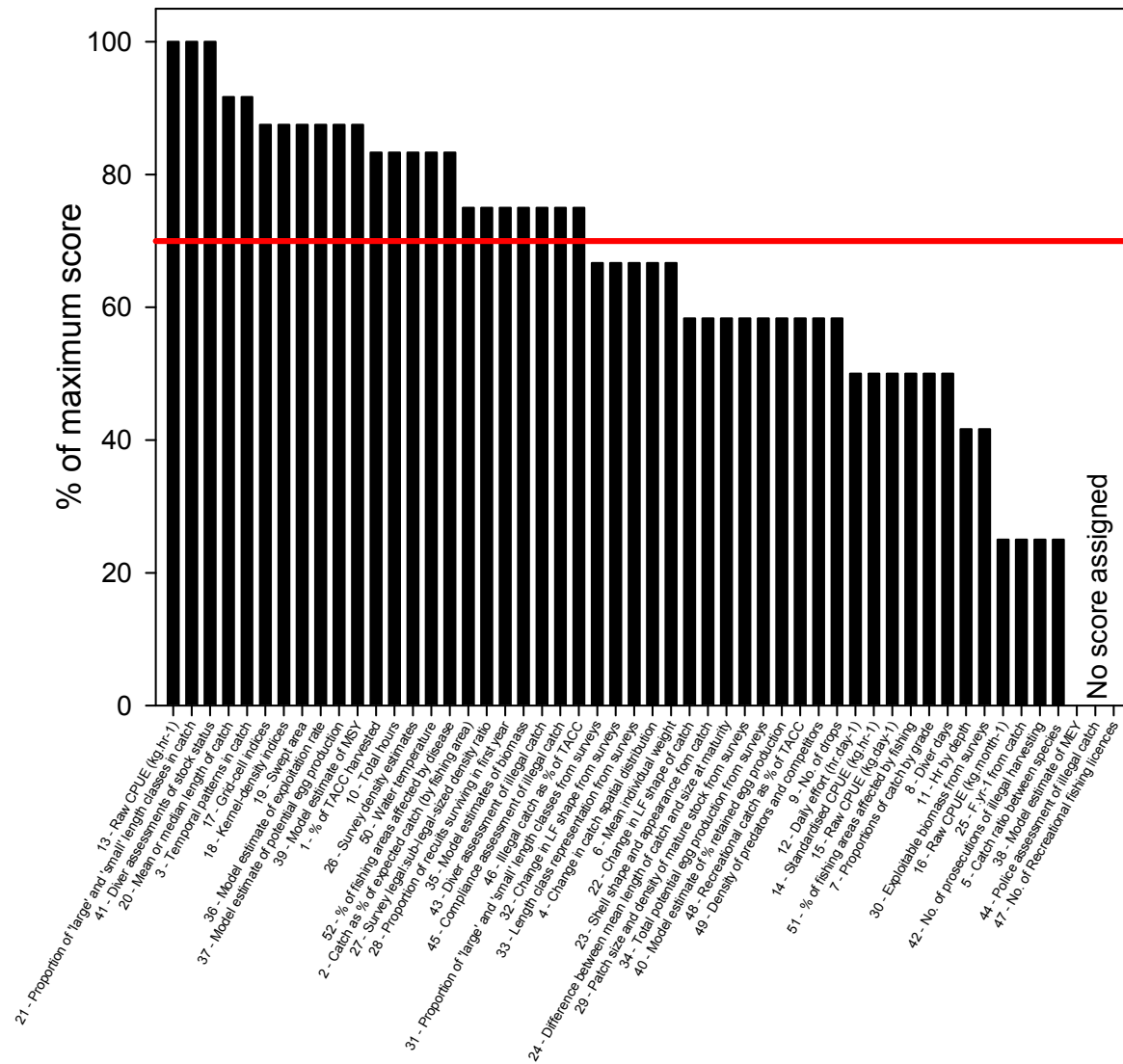


Figure 8. Percent of the maximum score determined for each biological performance indicator from the expert-panel workshops.

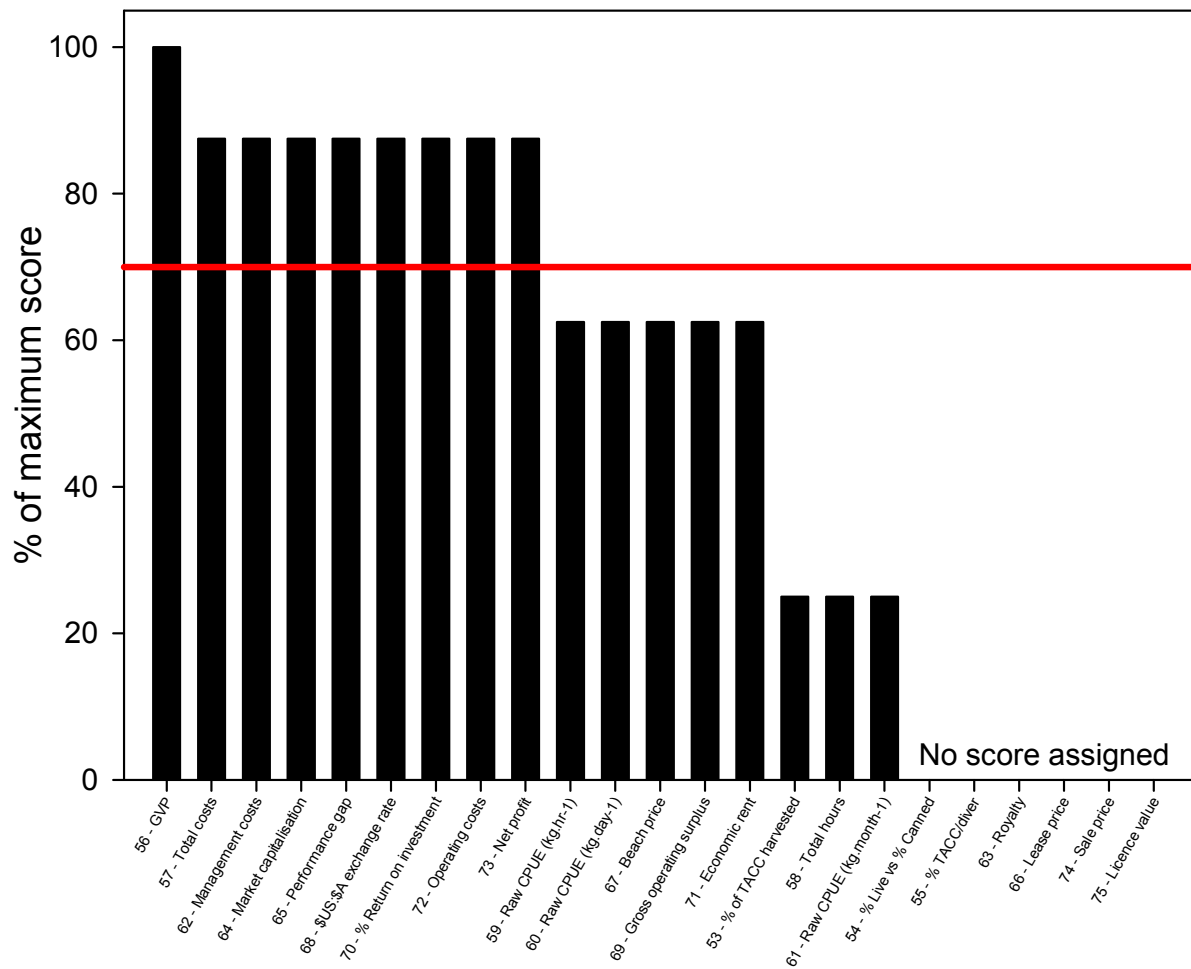


Figure 9. Percent of the maximum score determined for each economic performance indicator from the expert-panel workshops.

14.2.4 Discussion

International agreements (UNCLOS 1982, FAO 1995, UNFSA 1995) and national and state legislation (CFA 1991, NSESD 1992, EPBC 1999, NPFB 1999, State Fisheries Acts) require fisheries to operate within a management system focused on biological and ecological sustainability. In pursuit of meeting these agreements, fisheries management is typically guided by a formal framework, such as a fishery management plan (MP), that provides guidance for management decision making, and achievement of ecologically sustainable development. Assessing performance of a fishery against objectives described in a MP requires specified PIs for each objective.

Assessment of current fishery performance against the objectives using the PIs allows fishery managers to match fishing intensity with stock status (CFHS 2007). Consequently, PIs provide objective criteria against which fishery performance can be assessed, and are a central component of a harvest strategy. Obtaining PIs that accurately measure changes in fishery performance is difficult and, despite similarities among many fisheries, different PIs are used to assess fishery performance in different places – even for the same or similar species. Exacerbated by legislative and operational differences among fisheries, these differences mean use of the same set of PIs is not necessarily appropriate, possible, or even desirable. As identified in Chapter 2, this is because PIs should (1) be directly relevant to each fishery; (2) be estimable with sufficient accuracy to form the basis of clear management actions; (3) reflect the biology of the species concerned and; (4) be agreed to by stakeholders.

Performance indicators must be robust and provide clear and timely indications of variation in abundance and/or population structure (biological PIs) or economic performance of the fishery (economic PIs). There is limited agreement in the use of PIs in abalone fisheries, with a diverse, poorly-understood suite used currently (Gorfine *et al.* 2001).

The expert-panel workshops undertaken in this study provide the first step to assessing the ability of the PIs used in the management of abalone fisheries, identified in Chapter 2, to act as an ‘early warning signal’ of decline, as an indicator of improving resource status, or as an index of sustainability for these fisheries. Such an approach is not unique (Samarco 2008, Southall *et al.* 2009, Zajicek *et al.* 2009) and, because the expert panels were typically comprised of all stakeholders (i.e. divers, licence holders, research scientists and fishery managers), there was a balanced, broad representation which contributed substantial experience to the assessment. Importantly, the diverse composition of the expert panels enabled an additional 18 PIs to be identified, each with the potential to benefit assessment of abalone fisheries. The process of assessment was also rapid, thereby facilitating a timely consideration of the PIs identified. However, expert-panel approaches are entirely qualitative and “opinion driven”, and can fail to consider the “achievability” or quantitative elements of some PIs. These approaches can also be biased, particularly when comments on individual PIs are provided by one sector (or individual) and do not represent a collective, considered response. This means individual PIs can receive a ranking (i.e. high or low) that is inconsistent with their historical or potential performance.

Most of the PIs evaluated by the expert panels in this study were related to assessment of fishery performance against biological objectives and, predominately, those relating to sustainability rather than ecosystem integrity. From 75 PIs ranked, just three of these – raw CPUE, proportions of large and small length classes in the catch and diver assess-

ments of stock status – were ranked VU at all three workshops. Each workshop also identified an additional four PIs (catch ratios between species, CPUE as $\text{kg}\cdot\text{month}^{-1}$, $\text{F}\cdot\text{yr}^{-1}$ and number of prosecutions as a measure of illegal catch) as NU. Thus, only ~10% of all PIs evaluated were ranked consistently among workshops. Variation was clearly apparent both within and among States, with these differences likely reflecting different histories, management arrangements and data availability among fisheries.

Determining an overall score for each PI provided a mechanism to synthesise the rankings of the three workshops to a single value. For the biological PIs, 23 achieved a rating of $\geq 75\%$ of the maximum possible score (**Table 9; Figure 8**) and, consequently, can be considered “preferred” PIs (**Table 10**). Twenty-six biological PIs, each with a rating $< 70\%$, were considered “non-preferred”. Interestingly, this group contained (1) three measures of population structure derived from fishery-independent surveys, which have the potential to provide a measure of future recruitment to the fishable stock, and (2) two measures of the size structure of the commercial catch, which also provide important information on stock status (Tarbath *et al.* 2005, Mayfield 2010). Consequently, given the limitations of the expert-panel approach outlined above, this ranking does not mean these PIs should be discarded. Rather, where possible they should also be incorporated into the more formal, quantitative testing phase employing management strategy evaluation (MSE; Chapters 5-7).

This approach was also useful for identifying “preferred” economic PIs (**Table 10**). GVP was ranked VU at workshops 1 and 3 (**Table 9**), and consequently had a rating of 100% (**Figure 8**). A further eight economic PIs had a rating $> 80\%$, which distinguished them from the others as “preferred”.

There were also several State/zone-specific PIs identified. These included proportions of catch by grade (Western Zone, SA) and five PIs primarily relevant to Tasmania. These five were % Live vs % Canned product, %TACC/diver, Royalty costs, police assessment of illegal catch and number of recreational fishing licences.

Overall, the expert panel approach has enabled the perceived relative value of 75 potential abalone PIs to be identified by stakeholders in the SA and Tasmanian abalone fisheries. Consequently, this rapid, but qualitative and opinion-driven process has, at least as a “first cut”, enabled these PIs to be broadly categorised as “preferred”, “non-preferred” and “State/zone-specific”. This information can now be used by stakeholders in the fishery, in conjunction with the MSE analyses (Appendix 4), to select a suite of PIs suitable for assessment of abalone fisheries in each State or zone. Importantly, the suite of PIs selected should be as diverse as possible, and avoid autocorrelations among PIs (i.e. each PI should measure performance of a unique component of the fishery). This will require consideration of existing management and legislative arrangements, research capacity to service assessment of fishery performance against the PIs and the availability of data.

Table 10. List of “preferred” biological and economic performance indicators based on the % of maximum score determined at the three expert-panel workshops. Reference number provides a link to **Table 9** and **Figure 8** and **Figure 9**.

Management Objective	Reference no.	Performance Indicator	% Max. score
B i o l o g i c a l	13	Raw CPUE (kg.hr ⁻¹)	100
	21	Proportion of 'large' and 'small' length classes in catch	100
	41	Diver assessments of stock status	100
	3	Mean or median length of catch	92
	20	Temporal patterns in catch	92
	17	Grid-cell indices	88
	18	Kernel-density indices	88
	19	Swept area	88
	36	Model estimate of exploitation rate	88
	37	Model estimate of potential egg production	88
	39	Model estimate of MSY	88
	1	% of TACC harvested	83
	10	Total hours	83
	26	Survey density estimates	83
	50	Water temperature	83
	52	% of fishing areas affected by disease	83
	2	Catch as % of expected catch (by fishing area)	75
	27	Survey legal:sub-legal-sized density ratio	75
	28	Proportion of recruits surviving in first year	75
	35	Model estimates of biomass	75
	43	Diver assessment of illegal catch	75
	45	Compliance assessment of illegal catch	75
	46	Illegal catch as % of TACC	75
E c o n o m i c	56	GVP	100
	57	Total costs	88
	62	Management costs	88
	64	Market capitalisation	88
	65	Performance gap	88
	68	\$US:\$A exchange rate	88
	70	% Return on investment	88
	72	Operating costs	88
73	Net profit	88	

14.3 Assessment of PI against known fishery performance

14.3.1 Introduction

Fisheries management is typically structured by formal frameworks, such as management plans, that facilitate assessment of fishery performance against prescribed biological, ecological, economic and social objectives. Performance indicators (PIs) are often used to guide decision making, as objective measures of fishery performance. For PIs to be effective they must be reliable and robust. Thus, they must provide timely indications of stock status by being sensitive to, and effective in, detecting changes in abalone abundance at spatial and temporal scales used in fishery management. This is usually achieved most effectively through a suite of PIs, each capturing different areas of fishery performance.

This section follows from the review of PIs used in abalone and other dive fisheries in Australia and around the world that provided a 'working-list' of 56 PIs (Section 14.1; **Table 8**) and the subsequent qualitative assessment of PIs by stakeholders that identified a 'short-list' of 23 biological and 9 economic PIs (Section 14.2; **Table 10**). The primary objective of this section is to undertake preliminary, quantitative assessments of PIs against

available data. In the Discussion, strengths and weaknesses in the application of the PIs investigated and alternate methods to increase the sensitivity and accuracy of those PIs and their application are outlined.

The preliminary, quantitative assessments of PIs presented in this section are undertaken using data from three South Australian abalone fishery datasets with known changes in abalone abundance through time. These datasets cover three species (blacklip: *H. rubra*; greenlip: *H. laevigata* and Roe's abalone :*H. roei*) and are available across broad spatial (100's m – 100's km) and temporal (dives within days – years) scales. Data on catch, effort and size structure of the commercial catch are available for all three fisheries. These data are used to evaluate PIs based on catch and effort (e.g. catch-per-unit-effort (CPUE $\text{kg}\cdot\text{hr}^{-1}$) and mean daily catch (MDC)) and commercial catch sampling data (e.g. mean and median size (mm shell length (SL)) and proportion of small and large abalone). For two, greenlip off Cowell and blacklip in Waterloo Bay, position (GPS) and depth data obtained using the AbTrack system developed under Fisheries Research and Development Corporation (FRDC) Project 2006/029 (Mundy 2010) are also available. These data are used to evaluate spatial PIs identified in Mundy (2010; Tables 4.1 and 4.2).

14.3.2 Methods

The effectiveness of 11 PIs (**Table 12**) at detecting declines in fishery performance (i.e. abalone abundance) was evaluated at multiple spatial and temporal scales using three datasets: (1) a commercial greenlip fishery off Cowell in the Central Zone (CZ) of the South Australian Abalone Fishery (SAAF; Section 4.2.1; hereafter termed Cowell); (2) a fish-down (i.e. managed depletion of legal-size abalone, in excess of long-term sustainable limits) of blacklip in Waterloo Bay in the Western Zone (WZ) of the SAAF (Section 4.2.1; hereafter termed Waterloo Bay) and; (3) an exploratory fishery for *H. roei* in the WZ of the SAAF (Section 14.3; hereafter termed Roei). The three fisheries targeted different species under different fishing rules. Thus, each dataset is from discrete fishery, and shared few similarities.

Not all datasets provided data to assess all of the PIs listed and not all PIs 'short-listed' in Chapter 3 were evaluated due to limitations in the available data (**Table 12**). Estimates of all 11 PIs were obtained for Cowell and Waterloo Bay (**Table 12**). For Roei, analyses were limited to three PIs (CPUE, MDC and mean size; **Table 12**).

14.3.2.1 Description of the fisheries

Commercial fishery – greenlip, Cowell

The Cowell fishery is a separately managed fishery with its own TACC (Mayfield *et al.* 2008b). Fishery-dependent data were collected, by fishers completing a research logbook, detailing their catch and effort for each fishing day and submitting those data to SARDI Aquatic Sciences at the end of each month from 2006 to 2009. All greenlip harvested (minimum legal length (MLL): 130 mm SL) were 'shucked' at sea. Data on the length-frequency distribution of the commercial catch were either provided by fishers measuring their catch at sea using an electronic measuring board (during 2006 and 2007) or by fishers placing the shells in bags, labelled with licence and date details, that were later measured by SARDI using a standard measuring board. In 2006 and 2007, a small percentage (<5%) of the shell measurements were <130 mm SL and >180 mm SL. As shell lengths outside this range were likely to constitute errors in the use of the electronic measuring

board, data were truncated within this range. Position and depth data were collected via a GPS unit located on each vessel and each diver was required to dive with a Sensus Ultra® dive logger attached to their diving cage (after Mundy 2010). In 2009, a high proportion of the GPS data was not accompanied by depth data. Thus, GPS data where the vessel speed exceeded 4 knots were excluded and individual dive events were classified by the vessel operator manually indicating (i.e. pressing button on GPS logger) the beginning and end of a dive event.

Fish-down – blacklip, Waterloo Bay

The Waterloo Bay fishery was conducted over a total of 46 days, from 28 May to 12 July 2007 (SARDI unpublished data, Chick *et al.* in prep). Commercial fishing was restricted to two adjacent areas (termed Areas 1 and 2). Fishery-dependent data were recorded for each area. Total catch (kg, meat weight) was estimated by divers and validated by processors for each fisher-day and area. All blacklip harvested (MLL 130 mm SL) were ‘shucked’ at sea. The majority (>85%) of abalone shells harvested in each area were placed in bags by fishers, labelled with licence and date details and later measured (mm SL) by SARDI using a standard measuring board.

Licence holders and commercial fishers volunteered to participate in this fish-down. Each licence holder was allocated an equal proportion of the ~21 t (whole weight) TACC (derived from pre-fishing survey estimates of biomass and application of a harvest rule; 50% probability of the legal-biomass (whole weight) estimate at a 30% harvest fraction). If fishers stopped fishing before harvesting their allocated TACC, the remainder was redistributed to remaining willing fishers until the TACC was harvested. Position and depth data were collected via a GPS unit located on each vessel and each diver was required to dive with a Sensus Ultra® dive logger (after Mundy 2010). All data from Waterloo Bay were truncated to exclude records outside the prescribed fish-down areas.

Exploratory fishery - *Haliotis roei*, Western Zone

Between November 2000 and December 2002, WZ commercial abalone licence holders harvested *H. roei* (MLL 75 mm SL) from numerous small areas (typically <3 km of coastline), defined periodically throughout the 3-year project during six fishing periods (1 November to 30 December 2000, 1 February to 30 June 2001, 1 October to 30 December 2001, 1 February to 30 June 2002, 1 October to 30 December 2002 and 1 February to 30 June 2003) (Preece *et al.* 2004).

Fishers provided detailed information for each dive. This included fishing location (nearest named place), latitude and longitude at the dive entry point, time underwater and number of *H. roei* caught. The catch-weight data reported to PIRSA Fishwatch (the SA fisheries compliance agency) on the catch-disposal-record (CDR) were integrated with the catch (by number) and effort data obtained directly from the licence holders. This was done by apportioning the total of each day’s catch weight to each dive in proportion to the number of *H. roei* captured during that dive, on the assumption that the catch size frequency distribution and length-weight relationship were similar among the dives in each day (see Preece *et al.* 2004). Abalone shells (up to 300) from each dive event at each fishing location were placed in bags by fishers, labelled with licence and date details and later meas-

ured (mm SL) by SARDI using a standard measuring board. Only summary data describing the mean shell length were available for analyses.

14.3.2.2 *PIs from catch, effort and catch-sampling data*

PIs derived from catch, effort and commercial-catch-sampling data were analysed for all three fisheries. The PIs were catch-per-unit-effort (CPUE $\text{kg}\cdot\text{hr}^{-1}$), mean daily catch (MDC), mean and median size (mm shell length (SL)) and the proportion of small and large individuals harvested (<15 mm and >25 mm over the MLL, respectively; **Table 12**).

14.3.2.3 *PIs from GPS and depth data*

The position (GPS) and depth data obtained from individual fishers for Cowell and Waterloo Bay provided data at spatial scales relevant to that of fished populations from which two types of spatial PIs were derived and evaluated. The first, based on individual dive events are described using kernel density utilisation distributions (KUD; **Table 11**) for which the PIs examined were mean maximum distance (MMD), kernel utilisation distribution index (KDI) and mean corrected perimeter area ratio (PAC; **Table 11** & 4.2). Measures used to derive these PIs included mean area and mean perimeter values (**Table 11**). The second, based on dive location, are described by pooling data from individual dive events within defined spatial boundaries (e.g. hexagonal grids of 1 hectare; Grid cell measures; **Table 11**). PIs derived from these measures were frequency of cells fished per diver and the cumulative effort ratio (**Table 11** & 4.2). Other measures, that aid interpretation of these PIs, include single and multiple day use of cells, exploratory and active effort (**Table 11**).

14.3.2.4 *Data analysis*

The spatial and temporal scales across which the PIs were analysed varied among fisheries because of variability in data availability. Due to differences among fisheries and the lack of independence of data among the spatial and temporal scales within each dataset, data were analysed separately.

For Cowell, data were analysed at the scale of the whole fishery by year (**Table 12**). For Waterloo Bay, data were analysed at two spatial scales. These were Areas 1 and 2 separately and that of the entire fished area (Areas 1 and 2 combined), both at temporal scales of days and weeks. Spatial PIs were analysed for Areas 1 & 2 combined only at a weekly temporal scale (**Table 12**).

Roei data were only available as summary data, aggregated at the spatial scales of location (Port Neil), fishing area (Area 20C) and sub-zone (Sub-zone 1). Within each location, data were available at the temporal scales of days, weeks, months and 'fishing-period' (periods of 2-5 months where the fishery was open between November 2000 and June 2003 (see Section 4.2.1; **Table 12**).

Where data were available catch-per-unit-effort (CPUE, $\text{kg}\cdot\text{hr}^{-1}$; after Rice 1995), mean daily catch (MDC, kg), mean and median size (mm SL) of the commercial catch and proportions of small and large abalone harvested, were calculated for each of the spatial and

temporal scales specified above and identified in **Table 12**. Relationships between each PI and time were evaluated using simple linear regression analysis (after Zar 1996). CPUE (kg.hr⁻¹) data were log transformed and proportional measures of the size structure of the commercial catch were arc-sine transformed prior to analysis (Underwood 1997).

Table 11. The spatial indices of fishery performance used in these preliminary investigations, the method and spatial scale at which they are derived and a simple description of their interpretation.

Spatial scale	Metric	Method	Description of metric and its interpretation
Dive event	Mean maximum distance (length of KUD)	Greatest distance of the 90% isopleth for each individual dive event.	Shape metric describing the dive event (i.e. round or long narrow dive events), to be interpreted in combination with KUD area, KDI and PAC
	Mean area	Calculated area of 50% and/or 90% isopleths	Measure of area where 50% and 90% of fishing effort has occurred. To be interpreted in combination with KDI and PAC
	Mean perimeter	Calculated perimeter of the 90% isopleths	Measure of the perimeter length of dive events.
	Kernel utilisation distribution index (KDI)	Area of the 50% isopleth / area of the 90% isopleth	Upward trend indicates less concentrated fishing areas (i.e. less patchiness), downward trend indicates increase concentration of fishing activity (i.e. greater patchiness)
	Mean corrected perimeter area ratio (PAC)	$PAC = P / \sqrt{4 \times \pi \times A}$ P = perimeter pi = Pi (constant) A = area of polygon	An index of patchiness of fishing grounds. A PAC of 1 = circle, 1.1 = square, >1.1 = long convoluted shape. Trending downwards indicates reduced patchiness, trending upwards indicates increased patchiness
Location based	Freq. cells fished per diver	Total no. cells fished / no. divers active	Summarises no. of cells fished per diver over a temporal scale. This can be converted to m ² fished per diver.
	Single and multiple day use of cells	Total frequency of cells being accessed for 1 or more number of days over a given period of time	An index of revisitation to the same areas.
	Exploratory effort	Cumulative effort (min) in cells with <15 min of effort	Measure of low fishing effort
	Active effort	Cumulative effort (min) in cells with >15 min of effort	Measure of high fishing effort
	Cumulative effort ratio	Cumulative effort (min) in cells with <15 min of effort divided by cumulative effort (min) in cells with >15 min of effort	An index of concentrated fishing effort.

Evaluation of PIs to detect changes in fishery performance, reflective of decreases in abalone abundance, relies on the assumption that decreases did occur. There were three lines of evidence supporting this assumption. Firstly, for each fishery, the harvest strategies ensured large, commercial abalone harvests (Cowell (Mayfield *et al.* 2008b); Waterloo Bay (SARDI unpublished data, Chick *et al.* in prep.); Roei fishery (Preece *et al.* 2004)). Secondly, fishery-independent surveys conducted before and after commercial fishing provided evidence of substantial reductions in the abundance of legal-size abalone (see references above). Finally, anecdotal evidence from commercial fishers participating in the fishing stated that the abundance of legal-size abalone had been depleted to levels well below 'normal' working levels.

14.3.3 Results

Data quantity and quality differed among the three fisheries. For example, at the smallest spatial and shortest temporal scales data points irregularly represented single dive days, single diver days and single dives. Temporal variability of each PI reflected these levels of data quantity and quality. As expected, variability was lower when data were aggregated at the largest spatial and longest temporal scales.

There was little consistency in PI trends among fisheries (**Table 12** and **Figure 10 – Figure 20**). The most consistent changes in PIs through time were observed at Cowell (**Table**

12; and 4.2). In contrast, fewer PIs changed consistently in Waterloo Bay (**Table 12**; Figures 4.4 – 4.8), whilst for Roei the PIs displayed little temporal change (**Table 12**; Figures 4.9 – 4.11).

Numerous PIs changed through time in a manner consistent with declines in fishery performance and reflective of reductions in legal-size abalone abundance. These changes were apparent in three ways. First, PI estimates changed substantially through time. This was particularly evident for PIs at Cowell (**Figure 10**) and for mean size, median size, proportion small and proportion large in Waterloo Bay (Figure 4.5 and 4.6). In contrast, CPUE on *H. roei* increased steadily by fishing period in Area 20C (**Table 12**; Figure 4.9). Second, estimates of 12 PIs changed significantly through time (**Table 12**). Ten of these 12 were consistent with decreases in abalone abundance, but there were two exceptions: MDC increased significantly at a daily temporal scale in Area 2 of Waterloo Bay (**Table 12**; Figure 4.4) and CPUE on Roei increased in Area 20C across fishing periods (**Table 12**; Figure 4.9). Third, PIs derived from commercial-catch-sampling data (i.e. mean size, median size, proportion small and proportion large) displayed differences through time that were more consistent with decreases in fishery performance and reductions in legal-size abundance than those from CPUE and MDC.

PIs from Cowell provided the most consistent inference of fishery performance. This is because all of the PIs derived from catch, effort and commercial-catch-sampling data, along with several PIs based on GPS and depth data, displayed temporal patterns consistent with large reductions in harvestable biomass. Four (CPUE, MDC, mean size and KDI) of the 11 (36%) PIs examined changed significantly through time (**Table 12**; **Figure 10 & Figure 11**). Whilst clearly apparent, but non-significant, changes in the percent of small (MLL <145 mm SL), large (>155 mm SL) and median size of greenlip harvested, also reflected decreases in abalone abundance through time (**Figure 10**).

In contrast, PIs from the Roei fishery provided the least consistent evidence of a reduction in fishery performance. This is because most PIs displayed little change through time, especially CPUE and MDC (**Table 12**; Figures 4.9, 4.10 and 4.11). The exceptions to this pattern were the reductions in mean size at Port Neil where data were aggregated to weeks (**Table 12**; **Figure 10**) and the significant increase in CPUE in Area 20C among fishing periods (**Table 12**; Figure 4.9). This later trend is inconsistent with the reductions in *H. roei* abundance that occurred in this area.

In Waterloo Bay there were changes in PIs through time indicative of reductions in fishery performance. Three PIs (proportion small, KDI and effort ratio) displayed significant changes through time within Areas 1 and 2 combined that were consistent with reduced fishery performance (**Table 12**; Figures 4.6, 4.7 and 4.8) and the proportion of small abalone harvested increased significantly through time in Area 1 when data were aggregated by days (**Table 12**; Figure 4.6). PIs derived from commercial-catch-sampling data (mean size, median size, proportion of small and large abalone) generally displayed differences between the start and end of the fishing period that were consistent with decreases in fishery performance from reduced fishable biomass, but these changes were not significant (Figures 4.5 and 4.6). In contrast, CPUE and MDC did not show significant trends or patterns consistent with decreases in abalone abundance (**Table 12**; Figure 4.4). In fact, MDC in Area 2 displayed a significant increase through time, although this was likely an artifact of sea conditions limiting fisher access during the first 7-10 days of the fishery.

Table 12. Results of linear regression analyses (r^2 value; (number of observations)) for PIs from Cowell, Waterloo Bay and Roei, at temporal and spatial scales described in section 4.2. Analyses on CPUE ($\text{kg}\cdot\text{hr}^{-1}$), mean daily catch (MDC), and mean size (mm SL) were done on individual records where data were available. CPUE was $\log(x+1)$ transformed. Proportions were arc-sine transformed. Bold text represents statistical significance * $p < 0.05$; ** $p < 0.01$. Dash indicates no analysis. Reference number provides a link to tables and figures in section 3.

Data series	Performance indicator (reference no.)	CPUE	Mean daily	Mean	Median	Proportion	Proportion	MMD	Spatial - KUD		Spatial - Grid Cell	
		($\text{kg}\cdot\text{hr}^{-1}$) 13	catch 15	size 3	size 3	small 21	large 21	17	KDI 17	PAC 17	Fq. cells/diver 18	Effort ratio 18
Spatial scale												
Temporal scale												
Cowell												
Whole fishery	annual	0.173** (104)	0.129** (104)	0.039** (15663)	0.824 (3)	0.948 (3)	0.879 (3)	0.005 (343)	0.043** (343)	0.001 (343)	0.415 (4)	0.356 (4)
Waterloo Bay												
Area 1	day	0.004 (29)	0.019 (29)	<0.001 (6398)	0.075 (16)	0.296* (16)	0.026 (16)	-	-	-	-	-
	week	0.004 (29)	0.012 (29)	<0.001 (6398)	0.117 (6)	0.323 (6)	0.075 (6)	-	-	-	-	-
Area 2	day	0.010 (30)	0.168* (30)	<0.001 (4997)	0.010 (17)	0.034 (17)	0.002 (17)	-	-	-	-	-
	week	0.007 (30)	0.114 (30)	<0.001 (4997)	0.213 (4)	0.321 (4)	0.012 (4)	-	-	-	-	-
Area 1 & 2	day	0.002 (39)	0.007 (39)	<0.001 (11395)	0.075 (23)	0.210* (23)	0.037 (23)	-	-	-	-	-
	week	<0.001 (39)	0.002 (39)	<0.001 (11395)	0.551 (7)	0.607* (7)	0.291 (7)	0.004 (166)	0.149** (166)	0.009 (166)	0.42 (7)	0.886** (7)
Roei												
Port Neill	day	0.029 (42)	0.017 (43)	0.134 (41)	-	-	-	-	-	-	-	-
	week	0.057 (24)	0.008 (24)	0.323** (23)	-	-	-	-	-	-	-	-
	month	0.11 (11)	0.005 (12)	0.297 (12)	-	-	-	-	-	-	-	-
	fishing period (-b i-annual)	0.036 (6)	0.008 (6)	0.356 (5)	-	-	-	-	-	-	-	-
Area 20C	day	0.070 (70)	0.055 (70)	0.014 (68)	-	-	-	-	-	-	-	-
	week	0.094 (33)	0.064 (33)	<0.001 (32)	-	-	-	-	-	-	-	-
	month	0.100 (15)	0.065 (15)	0.022 (15)	-	-	-	-	-	-	-	-
	fishing period (-b i-annual)	0.864* (5)	0.430 (5)	0.330 (5)	-	-	-	-	-	-	-	-
Sub-zone 1	day	0.039 (116)	0.208 (116)	0.023 (109)	-	-	-	-	-	-	-	-
	week	0.067 (48)	0.039 (48)	0.032 (48)	-	-	-	-	-	-	-	-
	month	0.018 (17)	0.023 (17)	0.023 (16)	-	-	-	-	-	-	-	-
	fishing period (-b i-annual)	0.530 (5)	0.337 (5)	0.673 (5)	-	-	-	-	-	-	-	-

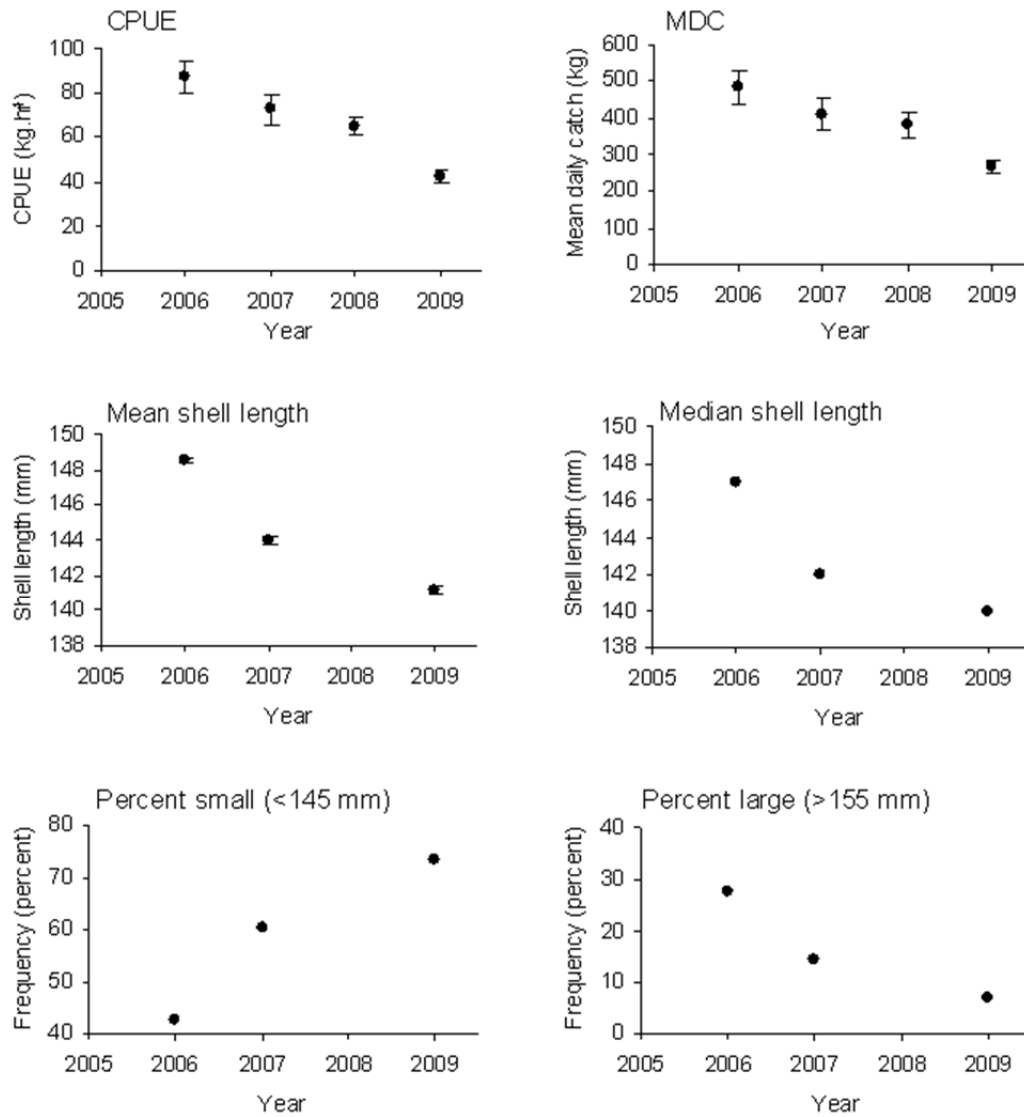


Figure 10. Mean CPUE (kg.hr⁻¹), daily catch (kg), shell length, median shell length, the percent of small (≤ 145 mm SL) and large (≥ 155 mm SL) commercially harvested greenlip from Cowell in 2006, 2007 and 2009.

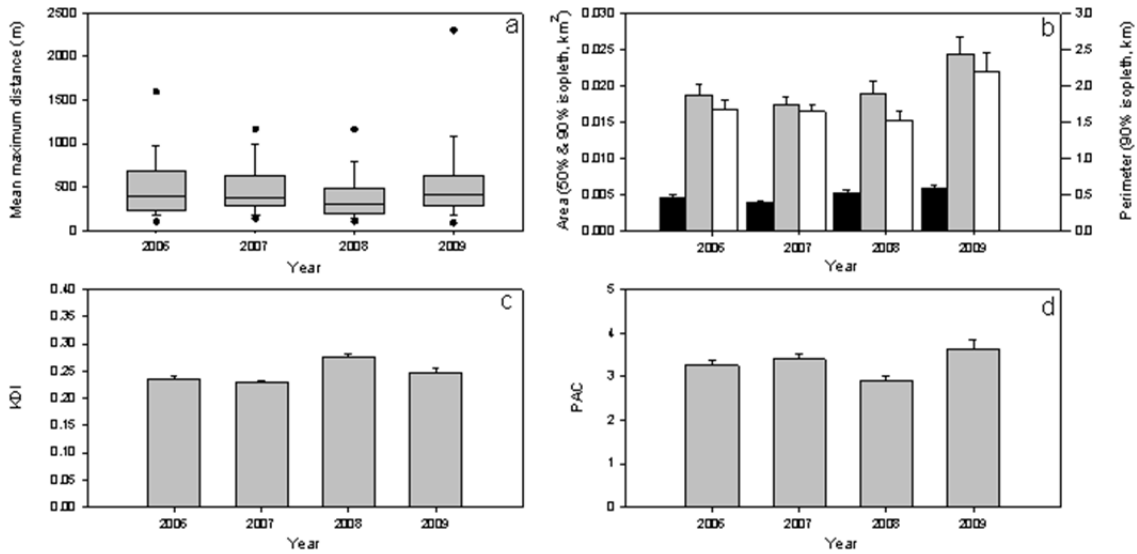


Figure 11. KUD metrics - Cowell. (a) Mean maximum distance from the 90% isopleth of individual dive events per year; median (black line), 25th and 75th quartiles (top and bottom bars, respectively), 5th and 95th percentiles (error bars) and outliers (dots). (b) Mean area of dive events per year using 50% (black) and 90% isopleths (grey) and perimeter of the 90% isopleth (white). (c) Mean KDI ratio per dive event per year. (d) mean PAC index per dive event per year.

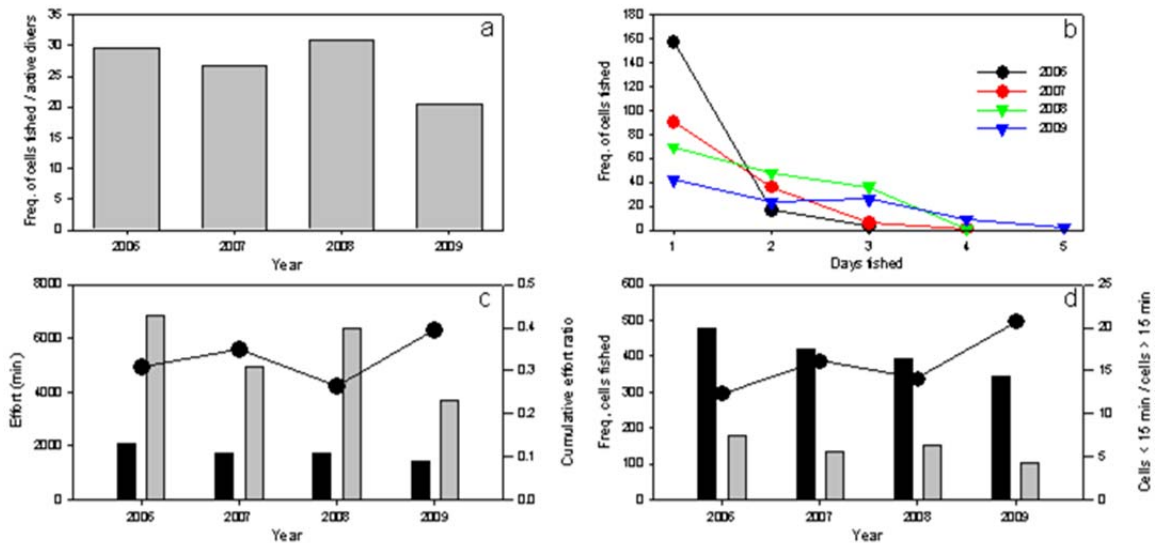


Figure 12. Location based metrics - Cowell. (a) Frequency of cells fished by per active diver per year. (b) Frequency of cells fished (>15 min effort) by all divers for 1 or more days per year. (c) Total effort (min) per year in cells with <15 min of effort (black bar) and cells with >15 min effort (grey bar) and cumulative effort ratio per year (total effort in cells with <15 min / total effort in cells with >15 min effort; dots and line). (d) Frequency of cells with <15 min of effort (black bar), frequency of cells with >15 min effort (grey bar) and ratio of the frequency of cells fished for <15 min effort / cells fished with >15 min effort (dots and line).

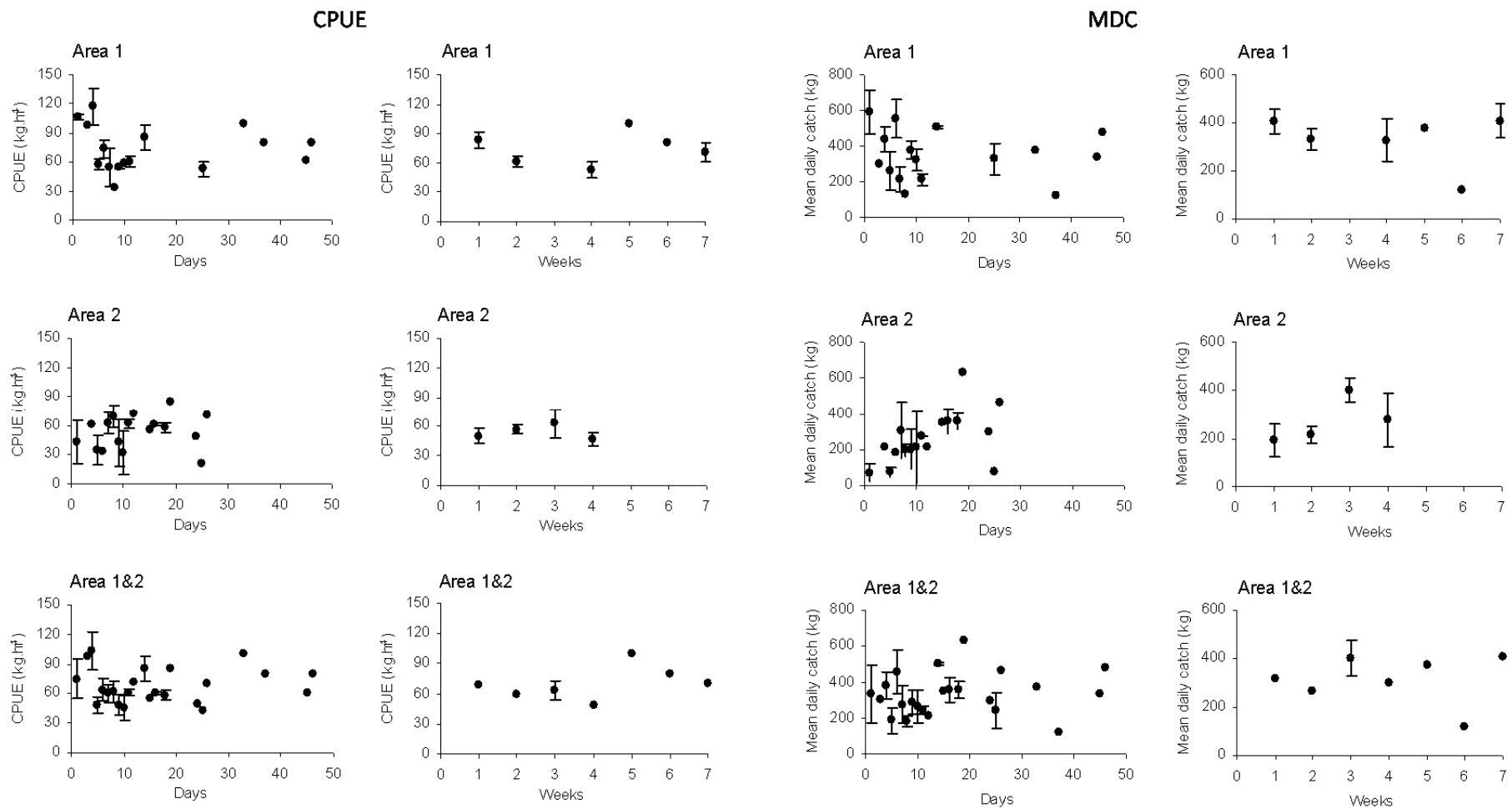


Figure 13. Mean CPUE (kg.hr⁻¹) and MDC (kg) of commercially harvested blacklip on days and weeks fished in Area 1, Area 2 and Areas 1&2 combined of Waterloo Bay.

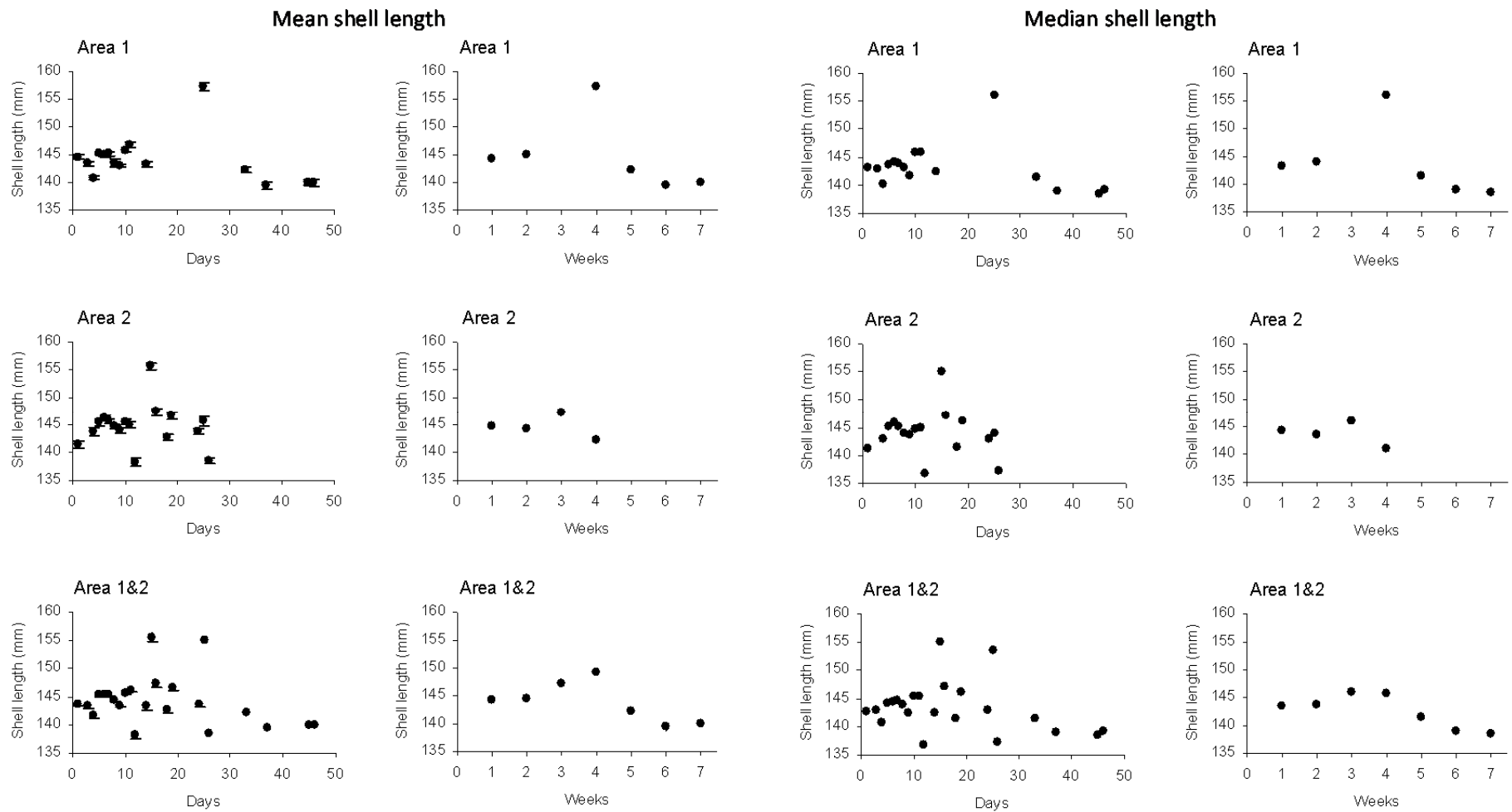


Figure 14. Mean and median shell length (mm) of commercially harvested blacklip on days and weeks fished in Area 1, Area 2 and Areas 1&2 combined of Waterloo Bay.

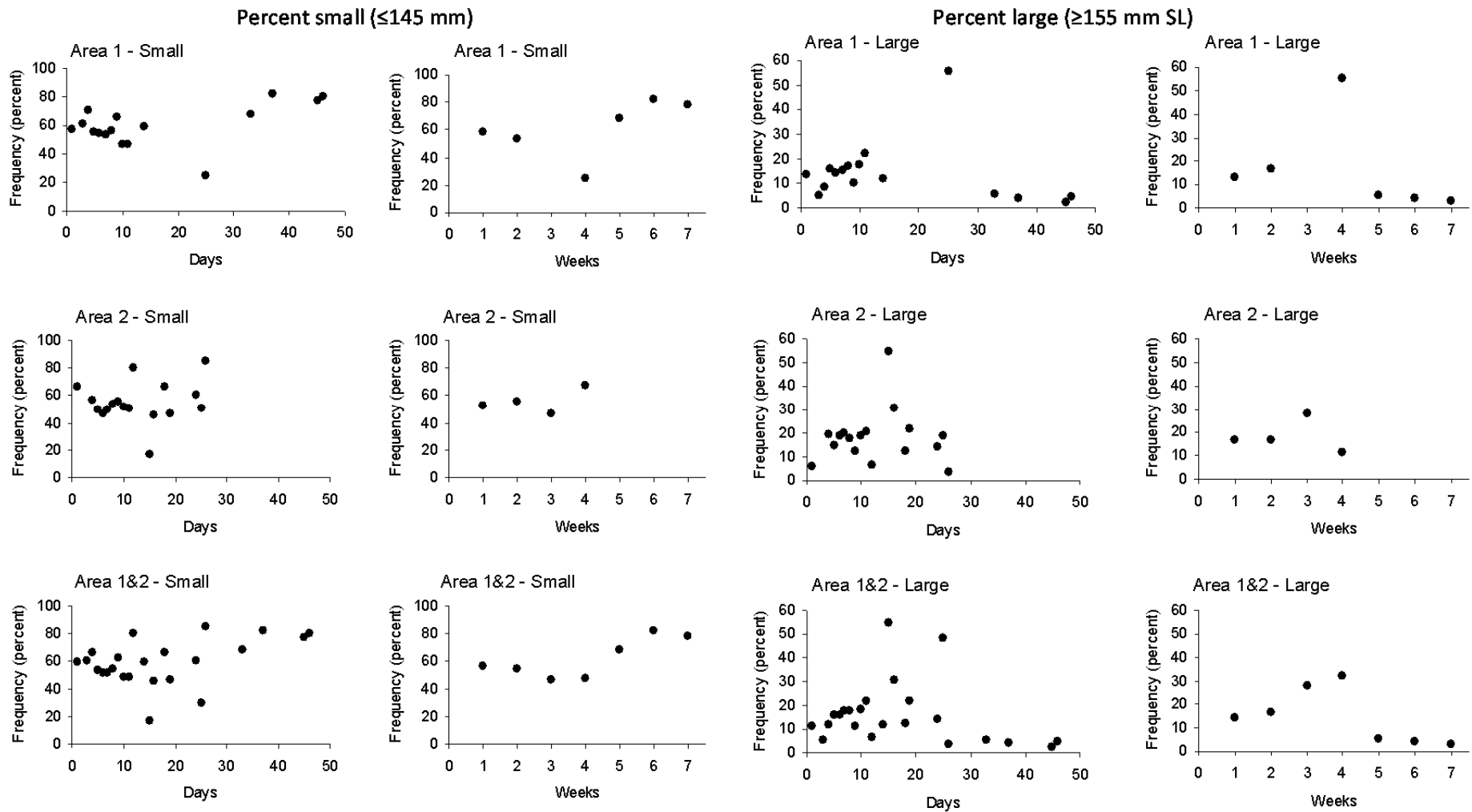


Figure 15. Percent of small (≤ 145 mm SL) and large (≥ 155 mm SL) blacklip commercially harvested on days and weeks fished in Area 1, Area 2 and Areas 1&2 combined of Waterloo Bay.

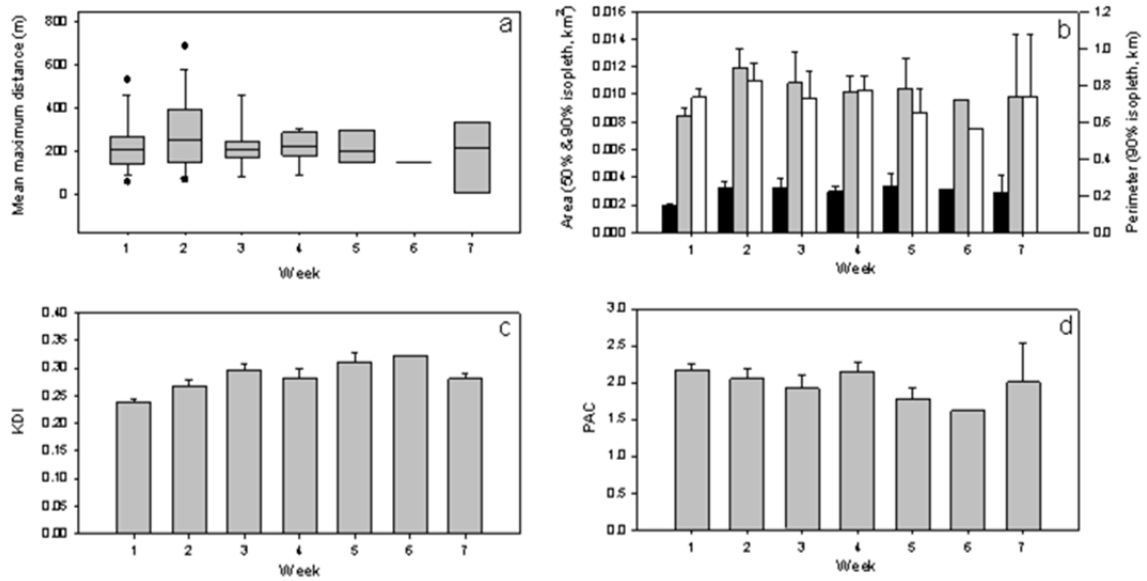


Figure 16. KUD metrics - Waterloo Bay. (a) Mean maximum distance from the 90% isopleth of individual dive events per week; median (black line), 25th and 75th quartiles (top and bottom bars, respectively), 5th and 95th percentiles (error bars) and outliers (dots). (b) Mean area of dive events per week using 50% (black) and 90% isopleths (grey) and perimeter of the 90% isopleth (white). (c) Mean KDI ratio per dive event per week. (d) mean PAC index per dive event per week.

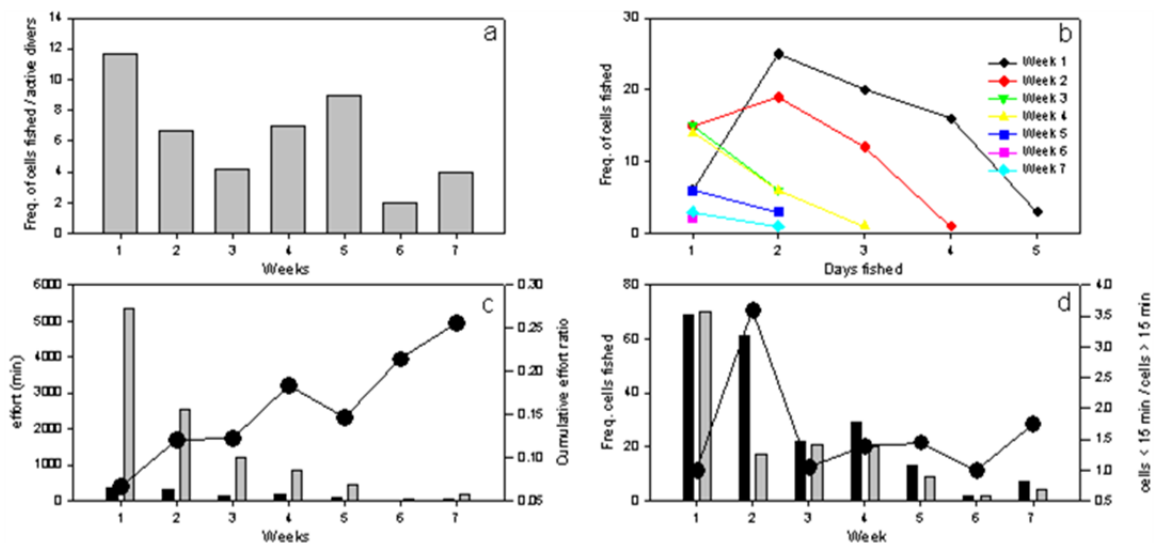


Figure 17. Location based metrics - Waterloo Bay. (a) Frequency of cells fished by per active diver per week. (b) Frequency of cells fished (>15 min effort) by all divers for 1 or more days per week. (c) Total effort (min) per week in cells with <15 min of effort (black bar) and cells with >15 min effort (grey bar) and cumulative effort ratio per week (total effort in cells with <15 min / total effort in cells with >15 min effort; dots and line). (d) Frequency of cells with <15 min of effort (black bar), frequency of cells with >15 min effort (grey bar) and ratio of the frequency of cells fished for <15 min effort / cells fished with >15 min effort (dots and line)

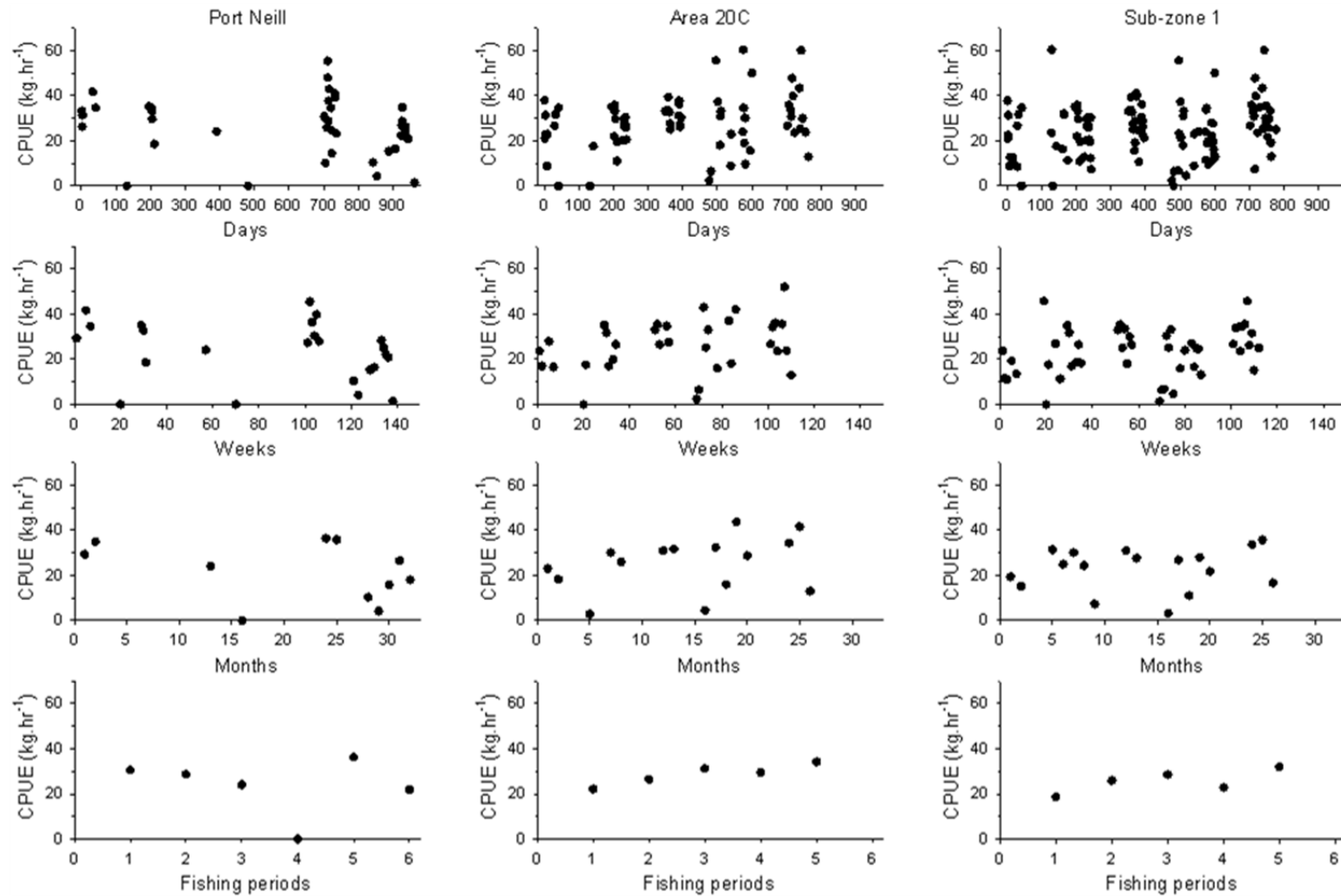


Figure 18. Mean CPUE ($\text{kg}\cdot\text{hr}^{-1}$) of commercially harvested *H. roei* on consecutive days, weeks, months and fishing periods from the commencement of fishing at three spatial scales, Port Neill (small), Area 20C (medium) and Sub-zone 1 (large), during the exploratory fishery for *H. roei* from 2000 to 2003.

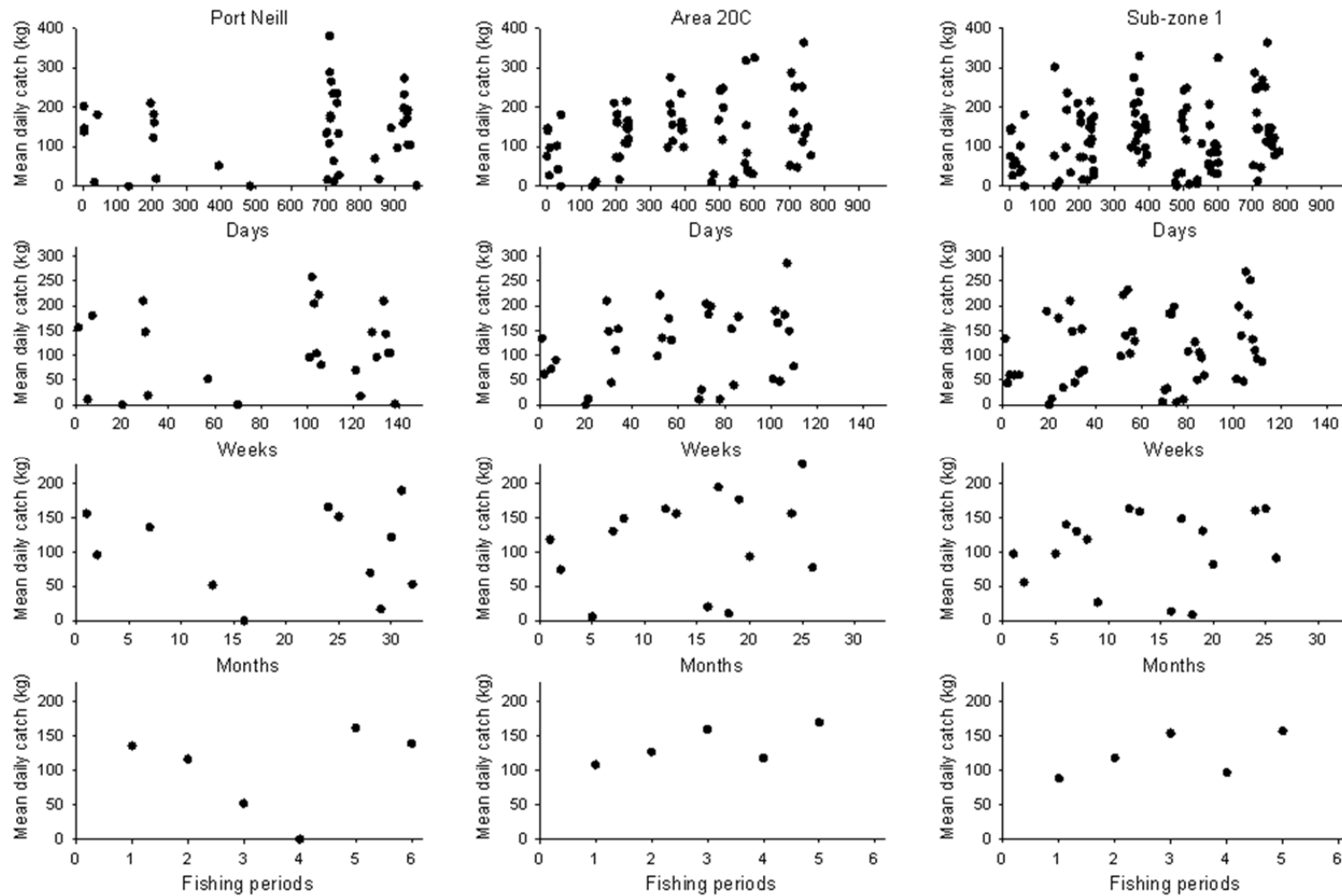


Figure 19. Mean daily catch (kg) of commercially harvested *H. roei* on consecutive days, weeks, months and fishing periods from the commencement of fishing at three spatial scales, Port Neill (small), Area 20C (medium) and Sub-zone 1 (large), during the exploratory fishery for *H. roei* from 2000 to 2003.

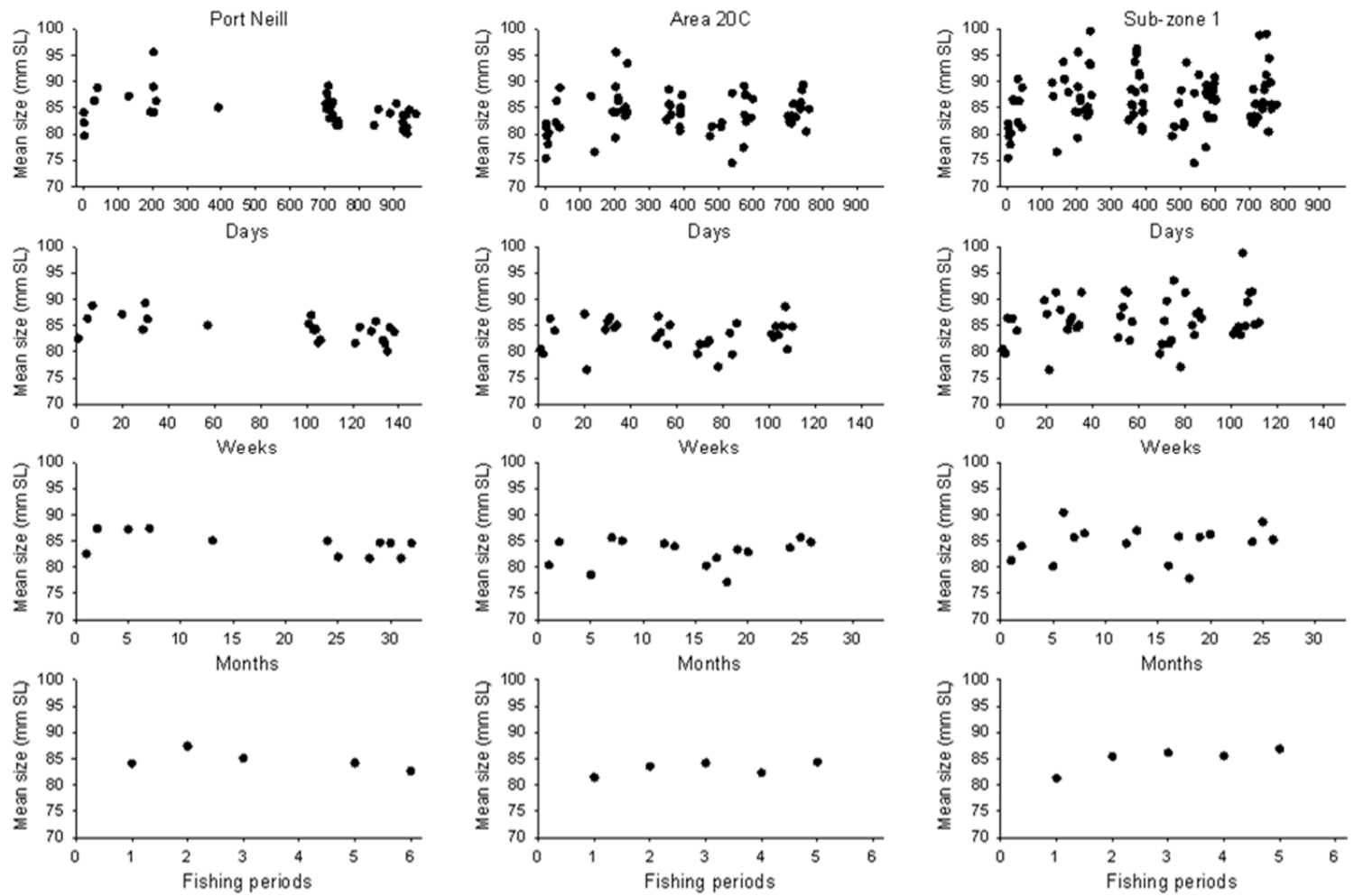


Figure 20. Mean size (mm SL) of commercially harvested *H. roei* on consecutive days, weeks, months and fishing periods from the commencement of fishing at three spatial scales, Port Neill (small), Area 20C (medium) and Sub-zone 1 (large), during the exploratory fishery for *H. roei* from 2000 to 2003.

14.3.4 Discussion

Measurement of fishery performance is an essential tool for managing fisheries and assessing them against fishery management objectives. PIs provide objective, quantitative measures by which to assess fishery performance, providing there is an understanding of the reliability and sensitivity of PIs at the spatial and temporal scales of management. This includes the integrity of the data used in their assessment (including decision rules to include or exclude data), the ability of each PI to measure change (appropriate analyses and power) and the influence of fisher behaviour and other biological, ecological, economic and social processes on PIs.

In this section, data from three fisheries were used to evaluate 11 candidate PIs. The data for Cowell were the only data available from a fishery operating commercially that had a consistent number of experienced fishers targeting a familiar species over an extended period of time (4 years). Whilst this fishery was initiated to allow harvest of a previously underexploited greenlip population, catches were not sustained and the fishery was closed after four years of fishing. Thus, the data for Cowell are most similar to those that would be expected from a fishery experiencing a rapid decline in stock levels. In contrast, (1) blacklip harvests from Waterloo Bay comprised a fish-down where stocks were depleted to levels below those normally experienced by fishers to test the efficacy of fishery-independent surveys on blacklip in SA and (2) the Roei fishery was designed to investigate the potential of a commercial fishery on this species in SA. Hence, the primary purpose of data collection for all these fisheries was not to test the performance of prospective PIs. Nevertheless, these data have been used to provide the preliminary analyses of PIs undertaken in this section. Consequently, it is likely that differences among fisheries contributed to the variable outcomes evident from the analyses undertaken.

Initial, quantitative application of the 11 PIs analysed highlighted marked changes in some measures of fishery performance. Notably, at Cowell, measures of CPUE, MDC and mean size displayed significant decreases through time. There were also prominent (but not significant) decreases in median shell size and the proportion of large shells, and an increase in the proportion of small shells in the commercial catch. Similarly, while few PIs showed significant change in Waterloo Bay, there were changes in the size structure of the commercial catch, as mean and median size and the proportion of small and large blacklip harvested varied through time. In particular, these measures provided substantial contrast in their initial and final values, consistent with expected declines in abundance. Thus, a key finding was that the consistency in the patterns observed across these fisheries in PIs derived from measurements of the size structure of the commercial catch suggests these PIs will provide a valuable measure of fishery performance. In contrast, the lack of consistency in measures of catch rate (CPUE and MDC) suggests these PI are less sensitive to changes in abalone abundance (Breen 1992, Prince & Shepherd 1992, Gorfine & Dixon 2001, Officer *et al.* 2001).

Given their recent development, interpretation of PIs derived from the position and depth data was less clear. However, the preliminary analysis of these spatial PIs indicates that changes in fisher behaviour – recorded as changes in the spatial distribution of fishing effort from GPS logger data – reflect declines in fishable stock similar to inferences made from more traditional PIs. For example, off Cowell and in Waterloo Bay, changes in the distribution of fisher effort within dive events through time (i.e. changes

in 50% and 90% isopleth area and associated increases in KDI) indicated that fishers were distributing their effort more broadly through time, from patches of high densities to those of reduced density and lower abundance. Whilst these patterns were consistent with those of changes in the size structure of the commercial catch and CPUE and MDC at Cowell, the limited understanding of the utility of these spatial PIs challenges their application and integration with more traditional measures of fishery performance.

Future assessments of abalone fisheries will require reconciliation and selection from among available traditional and novel spatial PIs to provide the most effective and informative suite of PIs with which to assess fishery performance, and hence stock status. The value of being able to select a diverse suite from a range of reliable PIs is that fisheries management can be based on greater certainty because the multiple PIs are derived from numerous independent data sources. Suites of PIs will also likely need tailoring to account for data availability, differences among species, spatial scales of assessment and management and importance of each area to the fishery. Thus, it is unlikely that each species/area combination will have the same set of PIs. For example, areas of high importance and/or high risk could be assessed using a broad, comprehensive suite of PIs, whereas areas of low importance and/or low risk may only require assessment with a substantially reduced set of PIs.

In this study, data availability and sample size was an issue that frequently restricted the power of the analyses. For example, in Waterloo Bay, fishers stopped fishing in the fish-down when their own CPUE approached levels below those achievable elsewhere in the fishery and at which they would not normally fish, despite prior agreement to meet the objectives of the fish-down. This meant that the number of fishers operating at the spatial and temporal scales assessed was lower than anticipated and there were fewer data representative of divers and diver-days. This likely had a greater effect on the assessment than would otherwise be the case in a 'normal' fishery where time and space constraints are absent. Further complication was added to the data presented in this section in the form of bias. For example, again from Waterloo Bay, poor weather conditions during the first 10 days of the fish-down limited fisher access to abalone in Area 2 and kept initial levels of CPUE and MDC low, yet this was not accounted for in the analyses. Also, in the case of Roei, fishers were harvesting an unfamiliar species in unfamiliar habitat (semi-intertidal). It is highly likely that initial catch, effort and catch rate data for this fishery were relatively low prior to fishers obtaining more knowledge on this species habit and therefore reporting data more meaningfully used to derive measures of fishery performance.

Unrepresentative sample sizes and outliers within data-series, as well as factors extraneous to the biology of fished populations (e.g. fisher behaviour and market forces), may result in biased data that can substantially influence analyses of PIs and outcomes of a fishery assessment process. Issues of sample size and bias are particularly relevant to abalone fisheries as it is becoming more widely accepted that management needs to be applied at smaller spatial scales, with greater biological relevance, because abalone stocks comprise numerous metapopulations that exist at spatial scales smaller than most current fishery management units (e.g. Zones or Regions; Prince 2005, Mayfield & Saunders 2008, Miller *et al.* 2009). Future management of these fisheries at smaller scales will further limit data representivity. These issues highlight the need for explicit consideration and evaluation of data available for assessment (Chen *et al.* 2003).

Decision rules for excluding data that bias measures of fishery performance have been used in the assessment of Australian abalone fisheries. For example, in the Tasmanian abalone fishery, catch and effort data beyond the most recent years (~10) are excluded from the process to standardise CPUE to avoid the effects of effort creep (Mundy *et al.* 2006). Similarly, in the NSW abalone fishery, the first 50 diver-days of a new fishers catch and effort data were excluded from analyses to standardise CPUE and assess associated PIs, to account for a period of learning prior to these data being deemed reliable (Worthington *et al.* 1999). In several Australian abalone fisheries there have also been recent trends of commercial fishers selectively fishing larger or smaller individuals to service market demands (Mundy *et al.* 2006, Chick *et al.* 2009). Fishing practices such as these can substantially influence commercial-catch-sampling data (e.g. mean size, proportion small and large) along with daily catch and effort. Consequently, this selective fishing can strongly influence PIs derived from these data, notably mean size, median size, proportion small, proportion large and catch rates. Unrepresentative sampling regimes can also bias assessments. For example, abalone demographic parameters and fishing patterns often vary over small spatial scales requiring commercial catch sampling designs to obtain representative and cost effective data (Andrew & Chen 1997, Burch *et al.* 2010). These data issues identify that assessments of fishery performance using PIs depends fundamentally on decision criteria that describe minimum requirements for the collection and inclusion of data and the documentation of factors leading to biases in those data.

Describing and applying appropriate statistical analyses to assess significant temporal change in PIs and ensuring that statistical outcomes are biologically and/or ecologically relevant is also an important basis for effective fishery management. A key weakness of this study was the simplistic approach of the statistical analyses done to assess the significance of change in PIs through time and, therefore, their ability to measure fishery performance. In many cases it was difficult to reconcile patterns in the data with the outcomes of the statistical analyses. Hence, it is likely that more sophisticated and specific analyses are likely to be required when using these PIs to assess fishery performance. This could include consideration of (1) testing for differences among point estimates (e.g. differences between minimum and maximum or initial and final values) and (2) non-linear changes in PIs through time (e.g. current year to reference year; 5-year running means).

The preliminary analyses of candidate PIs undertaken in this section have highlighted the strength of those based on commercial-catch-sampling data, along with the need to explicitly consider (1) the suite of PIs that most closely match management objectives; (2) sensitivity of PIs to detect change, particularly their ability to measure decreases in abundance prior to stock collapse; (3) minimum data requirements; (4) factors that bias data; and (5) statistical methods employed. This approach will be extended in subsequent sections by MSE, which provides a more complete and sophisticated approach to assessing the robustness of PIs and associated harvest strategies.

15 Appendix 4: Stage 2 – Objectives 3 – 6

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15.1 Testing Abalone Fishery Management Strategies

15.1.1 Introduction

Chapters 14 to 20 all relate to the production and initial application of a simulation framework for use in the management strategy evaluation of abalone management or harvest strategies. A fundamental difficulty with the production of such a framework is that abalone populations are generally spatially heterogeneous in their biological properties. Even populations that are geographically close together can be very different in their growth and reproductive capacity so their basic productivity can vary greatly. This means that, with even large amounts of biological data it is not generally possible to fit a stock assessment model directly to numerous populations at once; or at least not without making some very strong assumptions concerning the adequacy of averaging across any biological heterogeneity exhibited by different populations within a management area. The spatial structuring of abalone populations has large influences upon the dynamics of the collective stock and of the fishery that is imposed upon it. The current solution is not to attempt to fit a simulation model to a given abalone fishing zone but to condition the multiple populations within the model to have a similar range of properties, which is done by attributing properties to each simulated population from distributions of those properties taken from nature. Unfortunately, although there have been a large number of populations sampled (Tasmanian populations were used because that data was both extensive and easily available) the analysis of the full diversity of variation has still to be completed.

15.1.1.1 *The Review of Growth Models*

Any simulation model to be used for testing abalone harvest strategies needs to be able to represent the stock dynamics in both a plausible and defensible manner. The construction of the simulation framework therefore required a number of preparatory steps that facilitated generating the required plausible description. First the formal selection of appropriate mathematical sub-models to represent the various biological properties needed to be reviewed. This was straightforward for properties such as the size at emergence and the size at maturity because a classical sigmoidal curve generally provides the most acceptable representation of events. However, during and following the original development of this project (in 2006/2007) the review of growth led to a whole new growth model for abalone being developed. In addition, its validity was tested by determining whether it was preferable to classical curves used elsewhere (the von Bertalanffy and the Gompertz curves); these developments continued through into the initial years of the formal project once it finally began (Haddon et al, 2008; Helidoniotis et al, 2011; Helidoniotis and Haddon, 2012, 2013). A strong emphasis was given to characterizing growth in abalone because of its powerful influence over the population dynamics and productivity.

15.1.1.2 *Other Biological Variables*

Re-working the wide array of biological data sets relating to morphology, maturity, growth, and related fishery data was necessary to permit the conditioning of the simulation framework so that its similarity to real abalone fishing zones was both convincing and defensible. Given the scale of the work, for example, size at maturity curves were fitted to data taken from more than 250 populations (Helidoniotis and Haddon, 2009), such analyses could only occur once the database containing the information was re-structured into a fully functional database allowing the extraction of multiple records, which was done a short time before the start of this project; that restructuring was primarily organized by Dr Craig Mundy of TAFI.

The intent of the MSE tests was to translate the management strategies currently in use into more formal representations that are more amenable to simulation testing. That way the strength and weaknesses of many of the assumptions in the present methods for managing the stocks could be tested and highlighted. If improvements or flaws could be developed or detected these too would be described.

This first chapter in Appendix 4 provides more details on the structure behind the analyses and research undertaken as a way of introducing the context in which the work was done as well as introducing the ideas behind management strategy evaluation.

15.2 Current Management in Abalone Fisheries

15.2.1 The Early Development of the Tasmanian Abalone Fishery

The Tasmanian commercial black lip fishery has operated continuously since 1962 (Tarbath et al. 2006). The fishery is a dive based fishery operating along the shallow coastal waters to depths of 35 m off the east and west of Tasmania including Bass Strait, the Kent Island Groups and King and Flinders Islands. As with other abalone fisheries, harvesting methods remain relatively unchanged relying on simple techniques with most divers operating from the surface using air pumps and hookah diver gear. Abalone are prized from or hooked off exposed surfaces of the substrata to which they are attached (Cox 1962, Tegner et al. 1989, Tegner 1993). By 1965, 21 divers were operating in the fishery. Limited licensing was introduced in 1968 where licenses were only issued to those that already held a license the previous year. The initial number of dive licences allocated was 120. An additional 5 licences were issued in the Bass Strait region of the fishery bringing the total number of dive licenses to 125, although the number of active licences now contributing the majority of the catch in each year tends to be rather less than that

The commencement of the fishery is taken from the point of view of the implementation of the first management controls in 1962 (see appendices in Tarbath and Gardner, 2012, for details of the history of the fishery), when a legal minimum length (LML) regulation of 127mm was enforced state wide (**Figure 21**). The use of LML are largely aimed at preventing the capture of immature fish based on qualitative assessments and ensured both conservation of the resource and yield-per-recruit. Fortunately the Tasmanian abalone scientific program was established before there had been any significant fishery (Harrison 1969, Harrison & Grant 1971, Harrison 1983) and to some degree regulatory controls were at least partially informed by scientific studies. The LML was determined

after the first tagging studies took place at Maria Island at Hopground Beach to estimate growth rate and establish an LML. This led to the development of a legal minimum length of 127mm (5 inches) which was the main form of management in 1962. An initiative to raise the LML to 152mm (about 6 inches) in 1964 was reversed in 1965 as having too great an impact on catch rates and fishing grounds. It was not until 1975 that studies on size at maturity were able to confirm that this legal size was protecting immature and recently mature biomass in some areas, however, further studies in other regions of Tasmania later revealed that 127mm is too low being below the size at maturity, particularly for populations along the south-west coast. Although the size varied through time the LML remained uniform throughout the State for 28 year between 1962 – 1989 along with no limits on allowable catch and both of these had implications for fishing pressure. For the west coast in particular the LML was too low and by 1984 it was universally accepted across the abalone industry that the Tasmanian resource was in a badly depleted state. From 1987 onwards the LML introduced were either larger or customized to particular areas (**Figure 21**).

Recorded catches in logbooks from the Tasmanian fishery date back to 1978, however, more reliable recorded catches from the fishery date only from 1985, following the introduction of a quota system. Over the 35 year catch history recorded in logbooks (1978 – 2012) reported landings have varied between 2,041 and 4163 t (**Figure 22**); although prior to the introduction of quotas there were reports of up to 500 t of catches that were unreported in Tasmania, at least some of which was landed directly into Victoria. Following the introduction of quotas the catches were reduced from 4,163 t in 1984 to 2,076 t in 1989 (greenlip included), settling on 2,100 t from 1990 onwards (**Figure 23**).

Harvests in the blacklip fishery are managed by regulating input and output controls. Input controls include limited licences (effort control introduced in 1969) and many restrictions on where and when fishing may occur (pre- and post-dive reporting is necessary by divers). Output controls include: 1) Legal minimum length (LML- the oldest management control introduced in 1962) where abalone need to exceed a certain shell length size defined by the LML in order to be harvested and 2) and Total allowable catch (TAC- introduced as Individual Transferable Quotas in 1985) where upper limits are set to restrict the catch (TAC) in terms of yield and allocated, following zonation starting in 2000, to each zone which is assessed and reviewed on an annual basis. Furthermore, within each zone, local caps are applied to some statistical blocks. The block is closed to fishing for the remaining part of the year if the cumulative catch within a year reaches the cap threshold. The recorded catches are highly variable throughout the State (Tarbath and Gardner, 2012) and are not necessarily indicative of productivity given that divers are fishing under the constraint of the TAC in each zone.

Length frequencies of the catch are also obtained along with a description of commercial operations (Tarbath and Gardner, 2012).

Between 1985 and 2000 the main features in the management of the fishery have been changes within two management controls: the TAC and the LML. Since 2000 zonation was introduced to encourage more spatial management into the fishery (**Figure 23**). A summary of the major management control changes is:

1. introduction of zonation dividing the fishery into East, West, Northern zone, and Bass Strait,
2. changes in the LML in the fishery overall and then in different zones in different years (Figure 21),
3. adjustments to the TAC in the fishery overall and then within each zone.

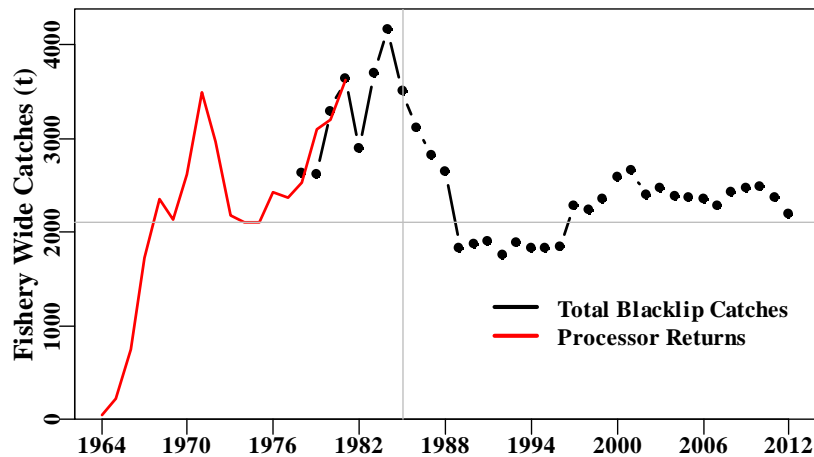


Figure 22. Recorded total annual catch (Tonnes) of abalone (blacklip) in Tasmania; peak catches were in 1984 where landings peaked at 4163.24 tonnes. Quotas were introduced in 1985 and quota reductions was enforced to reduce landings to a TAC of 2100T (including greenlip; horizontal grey line) over a 5 year period. Catches from Processor returns came from Tarbath and Gardner (2012)

Since 2000, the management the Tasmanian black-lip abalone fishery divided the 57 discrete statistical reporting blocks into management zones with a TAC for each of the five zones. Initially, in 2000 the Eastern and Western zones were established and the fishery was then further subdivided into the Northern Zone (2001) and Bass Strait Zone (2003). In addition, in 2000 the greenlip fishery was recognized with a separate TAC (**Figure 24**). Today there are 122 licensed commercial abalone divers and the TAC is made up of 3500 quota units with each unit having a set tonnage for each of the five zones. These quota units are distributed among approximately 600 quota owners operating under a range of input and output controls that are applied across the management range of the fishery.

The Tasmanian abalone fishery is described as a successful fishery (Hilborn et al. 2005) despite it being managed through a consideration of catches and CPUE, which is known to be a cause of abalone fishery collapses (Hilborn & Walters 1992) elsewhere. However, Mayfield et al. (2012) point out that abalone have been sustainably fished in Australia for 50 years. According to FAO statistics the Tasmanian fishery is the largest single managed abalone fishery in the world. This may not necessarily indicate best management practices in Australia as much as it suggests there has been relatively poor man-

agement in other abalone fisheries coupled with a denser human population elsewhere and a greater incidence of unregulated and unreported fishing.

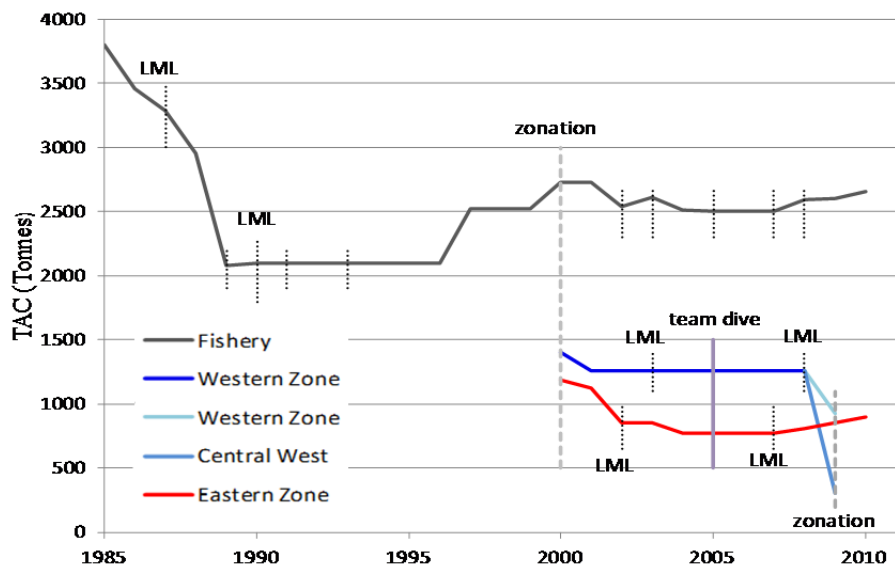


Figure 23. Management changes in the Tasmanian abalone fishery (greenlip and blacklip) including three major management controls; TAC, LML and zonation. Variation in the TAC is shown relative to the timelines of these and other management controls. The zone definitions are given in Tarbath and Gardner (2012). For a more detailed timeline of the Legal Minimum Lengths (LML, indicated by short dotted lines on the TAC line) refer to **Figure 21**.

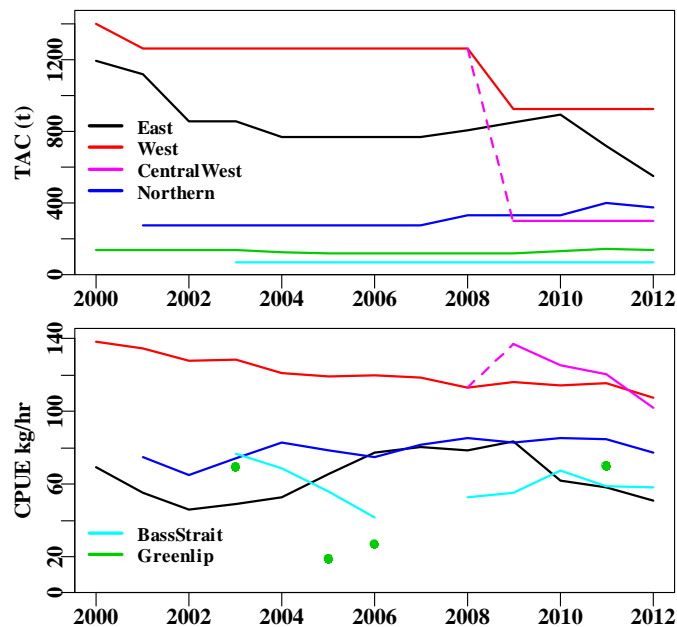


Figure 24. Changes in the Tasmanian abalone industry since the introduction of zonation in 2000, from data collected from log book records a) Total allowable catch (TAC), b) The related geometric mean Catch per Unit Effort (CPUE); the greenlip fishery is mixed with the blacklip fishery in the north and needs more detail in its analysis. Prior to 2008 the Central Western Zone was part of the Western Zone and therefore the Catch and CPUE estimates between 2000-2008 include both zones.

15.2.2 Tasmanian Weight of Evidence Approach

The present Tasmanian weight-of-evidence approach has no explicit performance measure targets or limits, although there appear to be some consideration of CPUE when making decisions but catch levels by area also appears to be important. This approach, is based on an informal synthesis of just a few quantitative performance measures (the spatial distribution of catches and catch rates along with the nominal level of catch rates and to a lesser degree the trends in catch rates, finally, any trends in commercial catch length frequencies are also considered although only very informally). The operation of this approach involves considering, for each statistical reporting block or area, the previous catches, the recent catch rates, and any length frequency data available along with diver observations. The prospective catch that could be expected from the area is agreed and once a complete zone has been discussed the total prospective catches are summed to generate a suggested TAC. Recent attempts to place realistic limits on prospective catches have involved tabulating the previous catch levels that have been taken from areas since 1985 and limiting prospective catches to a maximum of the upper 95th percentile. The implicit objective appears to be one of maximizing the catch that can be taken without compromising future catches.

Attempts to apply formal stock assessment model based management strategies has had mixed success (Victoria and NSW have used these, as have New Zealand, although two formal stock assessments were recently rejected there). Formal stock assessment models attempt to model the dynamics of the underlying stock and, for this reason, cannot handle unpredictable events such as disease or mass mortality events, until well after the events, if ever. Attempts have been made to implement such a model in Tasmania (Haddon, 2009; 2010), but while these were considered they were not adopted nor formed part of the assessment process. Importantly, such formal methods require explicit and operational management objectives (see Section 0). Large scale survey methods appear uninformative (Mundy *et al.*, 2005), although South Australia uses small scale surveys of their major fishing sites with some apparent success. The potential lack of representativeness of fishery dependent and independent data on length frequency, density, growth, size at maturity, and related variables puts increased doubt/uncertainty on the outputs from such formal assessment models.

Variants on the weight of evidence approach have been used in Victoria and NSW, although previously, formal stock assessment models were used to deliver stock assessments and generate management advice (Mayfield *et al.*, 2012).

15.2.3 South Australia's Formal Control Rules

The newer empirical harvest control rules as currently implemented in SA and which are being developed in Tasmania constitute a formal synthesis of the quantitative performance measures used in the weight-of-evidence management strategy. The formal synthesis allows for targets and limit reference points but the strengths and weakness of each approach still require discussion and testing. The weight-of-evidence approach has clearly managed to maintain a fishery for a very long time in each of the States where disease has not led to stock declines. However, as experiences in NSW have demonstrated, it does not always lead to optimum harvest strategies. It must also be realized that abalone stocks can slowly decline over decades; they do not always collapse rapidly. So there may well be an element of luck operating.

15.3 Management Strategies and their Evaluation

15.3.1 The Focus of the Current Work

The title of the original research proposal, *Identification and Evaluation of Biological Performance Indicators for Abalone Fisheries*, places emphasis on Performance Indicators as being the subject matter of the project work. However, even though when estimated as a series through time these can provide a measure of the stock status (performance) this only has value in the context of a management framework known as a management or harvest strategy. The objectives, listed in Section 4, which this part of the overall project are aimed at, provide a clearer notion of the subject matter being investigated than the overall title. Performance indicators or performance measures (PMs) are important components, especially when they are effective at reflecting dynamics changes in a fishery either in terms of its stock dynamics or the effectiveness of the divers or some other aspect of the dynamics influencing the fishery. However, once such effective PMs have been identified there is a need to determine how best to use them when trying to manage abalone fisheries. For that we use management strategies and those are the subjects of the remaining sections.

15.3.2 The Three Components of a Management Strategy

A management strategy has three components: 1) the data used to assess a fisheries stock, 2) the performance measures that are the output of an assessment of the fishery, and 3) the control rules that are used to translate the assessment outputs (the PMs) into specific management advice. The focus of this research is really about management strategies with all three components rather than solely upon the different measures of stock performance that can be used with abalone fisheries. This point is emphasized because of the confusion that arose when this project was being designed. At that time, despite attempts to clarify the underlying mechanics of management processes, emphasis was placed and focus put upon the identification of performance indicators (referred to as performance measures in the following sections). The intent was to discover which PMs exhibited contrast which reflected the relative abundance of abalone stocks through time. This is undoubtedly a vital first step, but would only be useful if combined with consideration of the harvest control rules or decision rules (HCR) that are used to translate a time series of PMs into management advice. What was needed to pursue the intent of the work was to examine full management strategies.

If any of the three components of a management strategy are changed then, strictly, this constitutes a novel management strategy. Thus, if we consider an abalone zone which is only managed using choices of LML and TAC, then if we elect to use standardized CPUE data, and use the gradient of changes in the CPUE over the last four years (years -5 to -1) as the fishery assessment, and then use a control rule that uses that gradient (from say year -3 to 0) to select a percentage change to the TAC from a defined scale (from say -20% to + 20%), then the three components combined would constitute a particular management strategy. If any of the major aspects of any of the components were then changed, for example using the gradient over the last six years instead of four (altering the assessment), or using a different defined scale of TAC percent changes (altering the control rule), then the result would be a different management strategy. The relative performance of such different management strategies can be compared in a simulation framework, and this is known as management strategy evaluation (MSE).

15.4 Terminology

Unfortunately, in fisheries science there can often be confusion over terminology, with different concepts being given the same name and the same concepts being given different names, and the field of MSE is no exception. For example a distinction is often made between performance indicators and performance measures. A performance indicator is generally identified as the statistic that is being estimated, which might be CPUE or proportion of mature animals in a sample, or some other measureable thing. It is generally implied that performance indicators differ from performance measures from the latter being a statistic that is compared with the same statistic through time or relative to some pre-defined limit and target reference points, thereby ‘measuring’ the performance of the stock or fishery. Such a distinction seems artificial in that without a comparative context any such statistic would not be useful so making the distinction between indicators and measures appears superfluous. In this work the two terms performance indicator and performance measure are used interchangeably, with a preference for performance measure as being more informative of the intent of the statistics estimated.

There is a similar confusion over management strategies and harvest strategies. The phrase harvest strategy has been used for a long time in fisheries science and its meaning has not remained static. The classical harvest strategies included constant fishing mortality, constant catch, and constant escapement (Hilborn and Walters, 1992), although there were variants and alternatives also available. The phrase harvest strategies sometimes still mean these but it can now also mean the same as management strategy.

The FAO Technical Guidelines for Responsible Fisheries series included the *Precautionary Approach to Capture Fisheries* (FAO, 1996) and *Fisheries Management* (FAO, 1997) appears to have been the first documents to have identified explicitly the need for targets, described as the desired outcomes for a fishery, and for operational constraints or limits, described as the undesirable outcomes that are to be avoided, and finally control rules which specify in advance what action should be taken when specified deviations from the operational targets and limits are observed. Early work on simulation testing of management arrangements (now known as management strategy or procedure evaluation; another potential confusion of terms) appears to have contributed to this approach to describing harvest or management strategies. Thus, FAO (1997) defines a *management procedure* as a description of the data to collect, how to analyze it, and how the analysis translates into actions. This is a standard way to describe a modern management or harvest strategy: define the data needed, the assessment of status, and the control rules used to generate management advice; however, in the FAO *guidelines* rather than the structural components, the importance ascribed to *management procedures* was placed on the investigation of how uncertainties influenced the management process (Butterworth & Bergh, 1993). Management procedures now appear to be restricted to the use of Monte Carlo methods to test the influence of uncertainties on a particular management and assessment framework fitted to a particular species. Management Strategy Evaluation, on the other hand, appears to be somewhat more general in that it encompasses any simulation framework that attempts to contain the stock dynamics and management arrangements for any fished species, which need not be as spe-

cifically defined as in a management procedure. However, there is a great deal of overlap between the two phrases and a distinction is not always clear.

While we can condition the simulation framework to have biological properties like a known abalone zone it is not possible to fit the models to an actual abalone fishery. Work is currently underway internationally (especially in the School of Fisheries, University of Washington, Seattle, and NMFS, Seattle) with respect to the development of generalized computational frameworks for conducting MSE work and the usage and meaning of particular terms has yet to become fixed or at least more generally accepted. In this work we will stick to the terminology of management strategy evaluation (MSE) using performance measures and control rules (alternatively called decision rules).

When the intent of the language used is understood then terminology becomes less important.

15.5 Management Strategy Evaluation

15.5.1 What is Management Strategy Evaluation

One reason that MSE work is relatively specialized is that the simulation framework used needs to be able to simulate the dynamics of the selected biological stock, the dynamics of the fishery imposed on the stock, the generation of simulated fishery data from the fishery, the stock assessment applied to that data and the control rule used to modify the present management options (generally changing the TAC), which are fed back into the dynamics of the stock in a feedback loop within the modelling framework (**Figure 25**). The feedback loop is an essential part of what makes a simulation a management strategy evaluation.

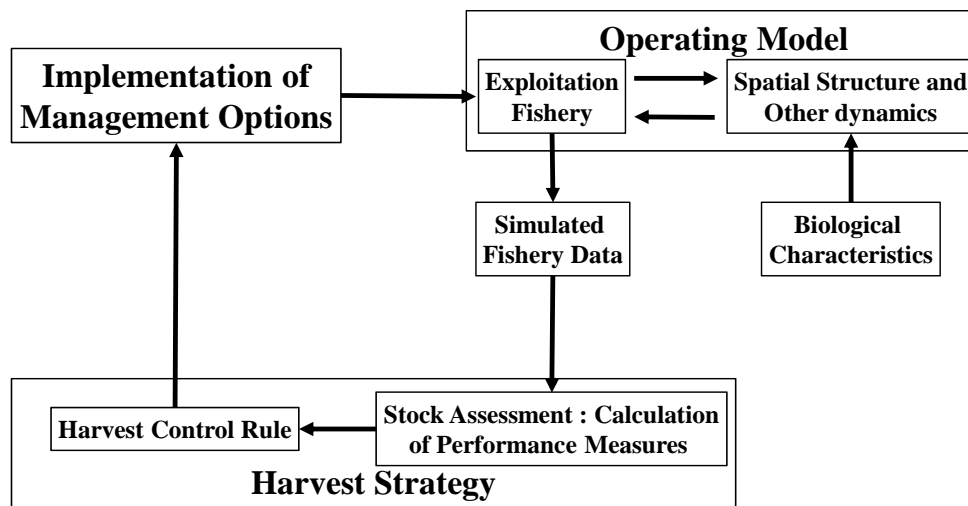


Figure 25. A diagrammatic representation of the main components of the Abalone MSE computational framework used in this project.

Because of the spatial complexity of real abalone stocks it is not possible to successfully fit the abalone MSE framework to the previously observed dynamics of an actual fished abalone zone. Instead, the biology of the populations simulated when generating a simulated zone can be conditioned on observed properties as seen in some real world abalo-

ne fishery (see the Operating Model methods below). This means we can only ever test the effectiveness of alternative management strategies upon simulated abalone zones that have biological properties that are only similar to known zones. By altering the recruitment dynamics within the framework we can also arrange to have the simulated zones have yields similar to those expressed in real abalone zones. However it remains impossible with the current information requirements to directly fit a simulated zone to the dynamics of a real zone.

15.5.2 The Components of the Abalone MSE Project

The analyses comprising the Abalone MSE work within the overarching project can be summarized diagrammatically as being composed of three components (**Figure 26**):

1. A characterization of the biology and fishery for abalone (needed to condition the simulation model on real fisheries).
2. An empirical exploration of recognized performance measures (such as trends in catches and catch rates, changes in the size distribution of abalone catches).
3. The simulation of realistic abalone fisheries (to permit the exploration of alternative management scenarios in the actual management strategy evaluation).

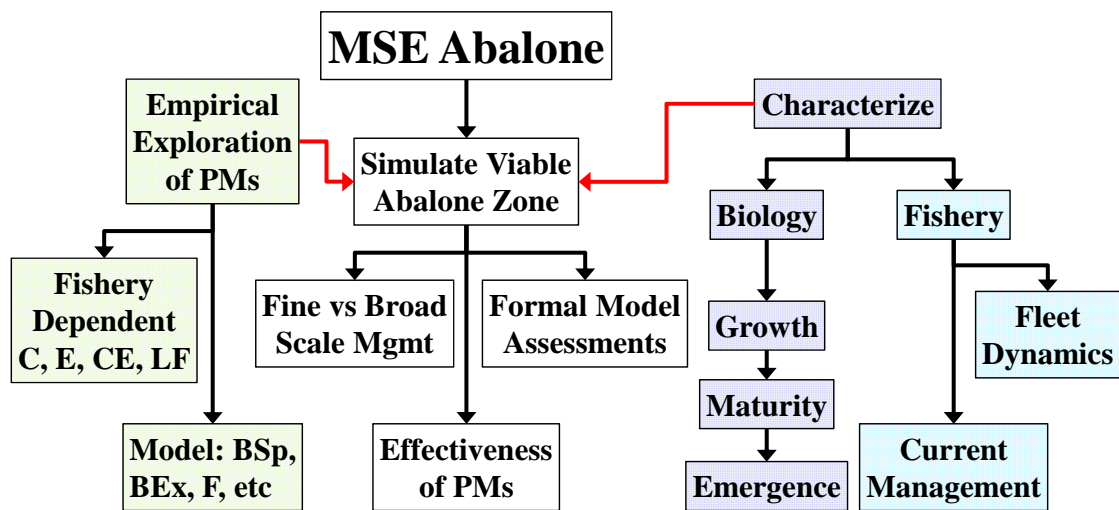


Figure 26. A diagrammatic representation of the three main components of the Abalone MSE program. The two outer pathways are needed to permit the construction of the main MSE simulation framework (the central pathway) and ensure that its output remains plausible and realistic.

Each of the three component pathways has their own data, methods, and results.

15.6 What Performance Measures are Available for Testing

The separate report by SARDI (Mayfield et al (2011), included here as section 13, has provided a detailed list of potential performance measures, however those which are currently being used to manage abalone fisheries within Australia include:

15.6.1 Fishery Dependent

- Catch
- Effort
- Catch per Unit Effort (CPUE)
- Distribution of effort and catch (spatial spread of effort/catch)
- Catch Length Structure
- Commercial business costs – economic data or proxies (the proxies might relate to the distribution of catches among divers, or the number of divers fishing in different geographical parts of a fishery).

15.6.2 Fishery Independent – Model Based

These tend to be based around model based estimates of parameters such as:

- Exploitable Biomass
- Mature Biomass
- Egg production
- Length structure of the stock (from surveys)
- Length structure of the catch (shed or industry sampling)
- Survey Indices of relative abundance

For many of these categories there are a number of performance measures that can be used. For example, Catches are used in all current fishery assessments but can be commercial, recreational or illegal, they can be spaced at different spatial scales, and can be distributed among different numbers of divers to mention just a few. The distribution of effort and subsequent catch is a particularly diverse category ranging from reef/population based catches to zonal based catches – reflecting the different scales of management that are possible. While there are many particular performance measures that can be included in each category, the categories above are the main groupings and these reflect the management levers currently and generally available (TAC, LML, and spatial management). The performance measures potentially available now that detailed spatial information on catches and effort are being collected using GPS data loggers cannot be included in this current project as such data has only just begun to be collected across the entire fishery in Tasmania during 2012 and onwards.

The stock assessment models that could be used include fully size-structured models which have been used elsewhere in management but have only been used in Tasmania to answer strategic questions. Alternatively much simpler surplus production models can be used and, given the surprising contrast in Tasmanian CPUE data, are remarkably effective in their ability to produce workable management advice (Haddon, 2011).

16 Fleet Dynamics

16.1 Introduction

The dynamics of a fishery consists of two main components; the stock and the fishing fleet. An understanding of the behavior of fishing fleets is therefore essential for achieving sustainable biological and economic goals. Failure to factor in the dynamics of a fishing fleet is considered a major weakness in fisheries management (Branch et al., 2006).

The term “fleet dynamics”, for a developed fishery, generally refers to how fishing fleets respond to management policies such as regulations, enforcement and closures (Branch et al., 2006). This includes: the locations where fishers operate and the investment and changes in fishing gear used (Branch et al., 2006). A problem exists when the management policies that are implemented, lead to unintended responses by fishing fleets. This mismatch in management intentions and fishery response makes up part of what is termed implementation uncertainty (Francis and Shotton 1997). The other aspect of implementation uncertainty, not considered here, is where there is an actual failure to implement the full intention of management advice or delaying when the implementation was intended (Dichmont et al, 2006; Haddon, 2011).

16.1.1 Management action affecting fleet dynamics

Generally, when new fisheries develop, management is very minimal and fishery behaviour is allowed to be entrepreneurial (Branch et al., 2006). In mature and developed fisheries, stock assessments become essential and management controls, designed as restrictions, are more prevalent.

Management controls can consist of both input and output controls so management decisions can include the introduction of limited entry, changes in total allowable catch, changes in legal minimum length (LML) and temporary closures (Branch et al., 2006). There can also be decisions to restrict season lengths, reduce and rationalize fleets, restrict a vessel’s capability (such as engine horsepower, tonnage and gear types), restrict the number of crew members on board, and the number of fishing licenses; although a need to avoid reducing fishing efficiency as a means of management, and thereby limiting profitability has been recognized (Branch et al., 2006). Such management decisions are made in response to perceived changes in performance measures which can relate to fishing mortality, spawning or available biomass, or their proxies (such as CPUE for relative available biomass).

The Tasmanian abalone fishery is a mature fishery with management controls that have developed progressively since its formal establishment in 1962/1963. The “fishing fleet” here is centered around the behaviour of divers rather than the behaviour of fishing vessels and therefore fleet dynamics relates to how divers pursue their catch, both spatially and temporally. This requires, at least, log book data consisting of daily records on how individual divers sub-divide their catches spatially.

The spatial structure of an abalone fishery is a major factor influencing both the space and time aspects of its dynamics. Any simulation of such a fishery requires a strategy for including how catches are removed across the spatial extent of the fishery, and this strategy should be informed by observations on how divers sub-divide their catches spatially.

At the same time, the economics of the fishery relate both to the value of the fishery as a whole but also the value to the individual fishers. To conduct a thorough analysis of the economics of being an abalone diver requires far more information regarding costs and revenue than is available in this study. Nevertheless, by examining the distribution of total catches among divers and how the divers sub-divide those catches among the blocks within a zone, some notion can be obtained of diver activities within a fishery and whether such fishing now permits the divers to make a living as commercial abalone divers.

Aspects of the abalone fishery which are generally unrestricted include fishing location and effort, whereas important limitations on the dynamics include the number of active divers each year; the number of dive licenses is restricted to 122 dive licenses but each year a far smaller number of divers catch the majority of the total catch. At the same time, the economics of the fishery relate both to the value of the fishery as a whole but also the value to the individual divers.

While there is a growing literature on fleet dynamics for many fisheries, the available material on the operation of abalone fisheries is very limited. Without an understanding of the dynamics of a fishing fleet, over-capitalization can occur where either too many operators join in on a fishery or the license to enter a fishery becomes overly inflated in value. Over capitalization may lead to the over exploitation of stocks, leading to depletion (Branch et al., 2006). In abalone fisheries this may manifest as a reluctance by the fishing industry to manage the TAC and the resistance to any decreases in the TAC relative to the readiness to accept increase since the TAC has often been observed.

16.1.2 A summary of responses against the various restrictions in input controls

16.1.2.1 Reduction in total allowable catch

Many divers are on a drip feed of quota, with quota holders trickling out quota units to fishers in small quantities; the assumption is usually that quota owners like to hold off while waiting to see if beach prices will improve. However, if only low levels of catch are made available the fishing equipment becomes too expensive to operate, so to maintain profits fishers reduce their costs by using less efficient fishing equipment and practices. The daily costs of fishing is reduced to keep the catch per cost outlaid (i.e. kg/\$) consistent, however the CPUE (kg/hr) may decrease as a result, which, as a side effect may produce a distorted view of the status of the fishery. The costs of fishing may also be reduced by fishing locally and reducing deckhands.

16.1.2.2 Spatially designed management controls

The fishery response is a re-allocation of effort by divers into distinct areas (Branch et al., 2006), which is clearly the outcome intended. It is however, difficult to predict ex-

actly where displaced effort will be moved to and so may have unintended consequences.

16.1.2.3 *Closed seasons which prevent year round fishing*

By having seasonal closures or placing a cap on catches from an area such that it is shut once the cap is reached can have the effect of the divers treating that cap as a competitive local TAC with a result that there is a race to fish. More divers may become active and each may increase their fishing effort by increasing the number of diving hours. With decreases in opening season, divers may also be more likely to risk operating in adverse weather conditions or by diving deeper and this potentially increases any risks to safety involved (Branch et al., 2006).

16.1.2.4 *Individual transferable quotas*

This may cause fishermen to alter their behaviour in a positive way to maximize the chance of profit and improve the value of their shares (Branch et al., 2006). The value of the ITQ share depends on the long term value of the fishery and quota owners may be committed to keeping the fishery sustainable (Branch et al., 2006). Quota owners have a strong bargaining power in the management of the fishery. They may either deliberately opt for a reduction in the TAC in order to increase stock size and eventually increase the value of their shares, or, if they are looking to sell their shares they may pushing to lower the LML, to increase the TAC and the CPUE, which will increase the value of their shares. Nevertheless ITQ's appear to be a control mechanism that may have positive affects and may mitigate any of the negative effects of other management controls (Branch et al., 2006).

16.1.2.5 *Market price*

This may encourage abalone divers to fish during seasons when festivals are common, which may put a lot of fishing pressure on a stock. This may have negative outcomes if such surges in fishing mortality occur during spawning periods (Branch et al., 2006).

16.1.2.6 *Stock declines*

Divers may respond by increasing fishing power through changing the manner in which they fish so as to maintain CPUE. Consequently, this can lead to at least a temporary oversupply of fish on the market which reduces the price paid to divers and processes.

16.2 Methods

16.2.1 The data

The dataset consist of daily catch effort data from commercial log books which consist of daily records of catch (tonnes) and effort (hours) for each individual diver, therefore a single record consists of the daily catch and daily effort for each individual diver at each location (recorded to sub-block accuracy). Although the fishery has been in operation since 1962 the most reliable data has been collected since 1985. Therefore comparisons at the block spatial scale can be made with 27 years of data between 1985 – 2011 inclusive. The fishery was rezoned in 2000, with further subdivisions over the next few years. Here we will only consider activities in the Eastern zone and what is presently known as the Western zone and the Central Western zone. This data has been reported

at the spatial scale of statistical reporting sub-blocks, and therefore analyses on sub-blocks consist of 12 years of historical data i.e 2000 – 2011. Zero data was removed, which means catches of 0 kg and effort of 0 hours were excluded in all analyses in order to log transform the data and fit a general linear model when standardizing catch rates; such records were uncommon however.

16.2.2 Spatial design

The spatial design of the analyses consists of five different spatial scales that are arranged in a hierarchical relationship (**Figure 27**). Like many fisheries the organization of the spatial design is unbalanced. That is, each unit within a spatial scale has a different area. Similarly each zone differs in area, has a different number of blocks or sub-blocks, and each block or sub-block will have different areas.

In addition the amount of data available differs. For example the number of records in one zone (the second largest spatial scale, (**Figure 27**), may be a fraction of that of another zone. Trends are likely to be less noisy as more data is available leading to a trade-off between information and variation i.e the signal to noise ratio. Where there are variable amounts of data between units within a spatial scale, comparisons can be less reliable due to the unbalance in the number of records. Therefore the amount of available data and number of records included needs to be considered when making comparisons. Similarly, more data is available on a larger spatial scale than on a smaller spatial scale and accordingly trends will be less noisy. It is therefore necessary to examine trends at different spatial scales when characterizing the dynamics of the diver behavior.

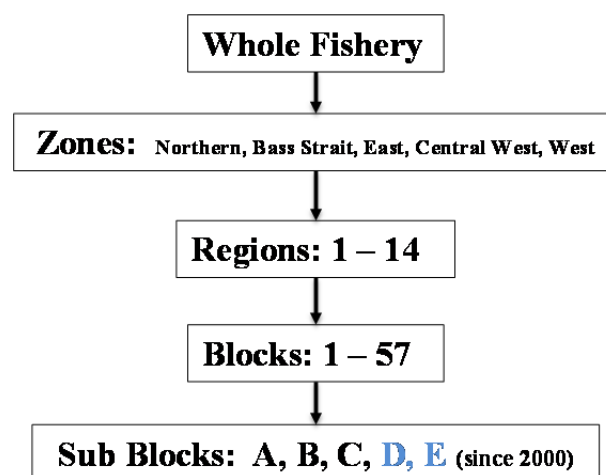


Figure 27. Spatial structure imposed on the Tasmanian abalone fishery. Zonation has only been in place since 2000, when sub-blocks were developed. The statistical reporting blocks have been in place since 1970, although reliable data is only available since 1985: the year that individual transferable quotas were introduced.

When considering catches, the sum of each level will equal the totals for each component of the level below. The sum of catches in each year in the sub-blocks of a block should equal the catches in that block for each year. However, this may not be the case when dealing with catch rates (either simple geometric or standardized) as there can be

interactions between spatial scales that would affect the outcomes. For example, if the catch rate trends in two of the zones were increasing and those in the other three decreasing, this could lead to the apparent catch rate for the whole fishery appearing to be flat and this might be thought to represent the average outcome across the fishery. Strictly, before a catch rate analysis of a given spatial level is undertaken attention should be paid to determining whether the catch rates in the component sub-levels are overly heterogeneous.

16.2.3 CPUE standardization

Apart from available biomass other factors may contribute to variability in CPUE between years. The four main factors for which data is available include the Year of catch, the Month of catch, Diver experience, and the statistical Block or Sub-block (Hilborn and Walters 1992). In addition, there may be significant interactions between Block and Month, or between Diver and Block, or possibly between Diver and Month. Average CPUE varies between divers and there is a turnover through time among the active divers. Changes in seasonality and the inter-annual distribution of effort among blocks also contribute to variation in CPUE (and the Month factor attempts to account for that). To account for the individual contribution of these separate factors a statistical standardisation of the CPUE data was conducted. The standardisation required daily records from individual divers. The CPUE data were normalised by using a log-transformation. The statistical models all had the form:

$$\text{Ln}(CPUE_i) = \alpha_0 + \alpha_1 x_{i,1} + \alpha_2 x_{i,2} + \sum_{j=3}^N \alpha_j x_{ij} \quad (1)$$

where $\text{Ln}(CPUE_i)$ is the natural logarithm of the CPUE (kg/h) for the i -th record, x_{ij} are the values of the explanatory variables or factors j for the i -th record and the α_j are the coefficients for the N factors j to be estimated (α_0 is the intercept, α_1 is the coefficient for the first factor, etc).

Five different log-linear models were fitted and compared in an effort to account for the effects of Year, Diver, Month of fishing, Statistical block, and any interactions between Block and Month. Interaction terms involving divers were all insignificant, but this may have been due to the limited amount of data from many of the divers in any single year leading to many empty cells in the analysis. All factors were treated as categorical. The optimum statistical model was selected on the basis of the Akaike's Information Criterion and after assessing the improvements to the adjusted R^2 after fitting each model.

16.2.4 Indices of Fishery Characteristics

The dynamics of the fishery may vary depending on the amount of available catch; a 3000 t fishery may differ dynamically to a 2000 t fishery. For example the number of active divers may vary, however the concentration of active divers with respect to the catch may be similar. Therefore this section deals how the dynamics may vary in the presence of management changes with respect to the available catch. Three indices are used based directly on biodiversity indices that are commonly used in the ecological literature. The indices used and their relation in a fisheries context are; richness which is

the concentration of divers with respect to the size of catch, diversity which is the number of effective divers (i.e those divers that land most of the catch), and evenness or the distribution of catch between divers.

Richness is a measure of the number of divers in relation to the size of the catch.

$$\text{Richness} = \text{diverconcentration} = d / C \quad (2)$$

Where d is the number of divers each year and C is the size of the catch (tonnes) each year..

The actual number of divers may differ from the effective number of divers and so an index of diversity is used to provide a measure. The diversity index is the interaction of both the number of divers and the distribution of catch among those divers.

$$H = \text{effectivedivers} = -\sum_{i=1}^R p_i \log p_i \quad (3)$$

Where p_i is the proportion of catch taken by the i th diver in the dataset; this statistic is taken from information theory. If all catch was equally distributed then this measure of diversity will be at its maximum and therefore the effective number of divers will equal the actual number of divers. The greater the number of divers and the more evenly the catch is distributed then the higher the diversity index.

The distribution of the catch refers to how evenly the catch is distributed among divers; the higher the distribution value the more evenly it is distributed.

$$\text{Evenness} = \text{distribution} = \frac{H}{\log(\text{divers})} \quad (4)$$

Where H is the diversity and divers is the number of divers operating annually.

16.3 Results

16.3.1 Spatial and temporal changes in catch relative to management changes

Two zones, namely the eastern and western zone, have the highest proportion of landings in the Tasmanian abalone fishery (**Table 13**). A trend of increasing catches in individual blocks from these areas has been reported in the recent history (2000-2011) compared to the earlier history prior to zonation (**Figure 28**).

Also, there has been a higher frequency of annual catches >250T (**Figure 29**). This frequency of high catches may be due to two things 1) effort creep, where the effectiveness of a constant amount of fishing effort has increased and 2) the introduction of team diving in 2005, where the catch of two divers is recorded as one. In the case of effort creep the number of hours spent diving will be a consideration. In the case of team diving this may appear as fewer divers operating in the recent history. During the most recent time period (2000 – 2011) the increase in catch was greatest in blocks from the western zone (**Figure 29**) and the decrease was greatest in Block 14.

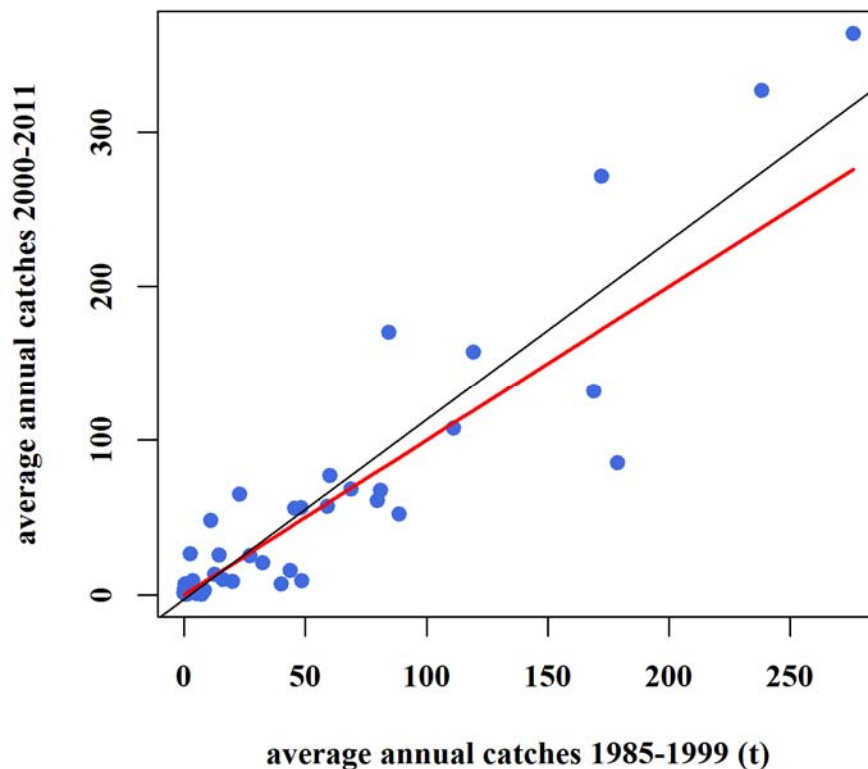


Figure 28. A comparison of the recent history of the fishery (2000 – 2011) to its earlier history (1985 – 1999) prior to zonation and beginning in 1985 which marks the first year of reliable records. Shown is the relationship between the average annual catches in each block over the period 2000 – 2011 and the average annual catches in the same block between 1985 – 1999. The grey line is the line of best fit of the relationship and the red line is the line of equality; 1:1.

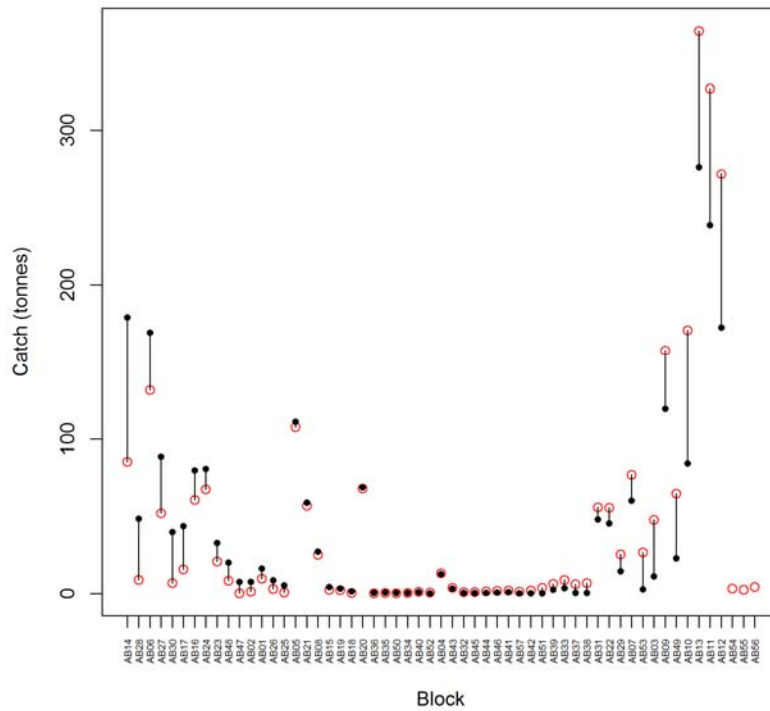


Figure 29. The difference in catch (tonnes) for each block between the two time periods: the recent history (2000 – 2011) in red open circles, and the earlier history (1985 – 1999) in black dots.

Table 13. Annual reported catches of Blacklip abalone by block, since the establishment of zonation and subblocks in the year 2000. NA implies no catches of Blacklip abalone were reported.

		2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	Mean annual total
Central West	AB05	45.42	117.24	105.46	73.23	57.73	78.93	100.18	88.95	162.69	170.74	133.55	156.49	341.3
	AB06	182.69	210.14	172.69	96.61	88.46	94.82	108.58	76.25	105.45	142.66	150.66	151.21	
	AB07	60.61	32.05	51.27	103.77	89.49	109.60	76.17	38.76	50.71	106.90	110.14	94.61	
	AB08	22.59	15.36	17.19	27.42	22.00	25.61	6.45	18.32	9.37	51.41	37.09	48.39	
Western	AB09	205.21	186.24	173.52	142.44	130.28	91.80	141.94	177.72	156.18	154.58	157.75	171.13	926.1
	AB10	147.54	151.98	142.51	238.49	181.44	148.83	197.76	231.20	178.25	110.26	157.81	158.88	
	AB11	326.45	312.26	359.75	344.45	373.85	388.92	383.85	353.82	345.04	243.72	243.56	246.72	
	AB12	281.55	289.95	235.94	229.21	249.56	310.58	228.30	267.32	305.14	327.12	276.64	256.21	
	AB13	434.94	366.65	389.25	356.27	316.61	246.63	272.50	323.05	419.54	417.89	409.93	414.82	
	AB14	97.28	157.47	100.59	116.23	103.71	90.43	83.67	70.41	56.38	62.79	69.59	15.33	
	AB15	1.65	2.92	0.58	2.06	7.45	8.48	3.37	0.27	0.93	1.27	1.00	0.21	
	AB16	71.28	108.00	72.15	59.28	49.75	55.99	66.88	55.98	63.87	51.98	38.75	36.90	
	AB17	28.61	19.56	15.68	17.24	20.01	19.66	13.15	8.19	8.03	20.18	9.91	7.82	
	AB18	NA	0.55	0.30	0.70	NA	NA	0.02	NA	NA	NA	0.01	NA	
Eastern	AB19	3.56	2.35	0.77	1.50	1.84	2.50	1.82	5.90	0.34	0.55	2.20	1.29	894.1
	AB20	61.99	56.41	62.02	87.78	92.46	115.66	73.42	68.21	49.94	51.00	70.52	30.24	
	AB21	60.04	49.51	57.73	54.15	52.16	62.32	65.89	63.47	61.39	52.16	59.12	47.13	
	AB22	69.20	40.49	46.26	35.42	35.21	35.53	70.66	61.28	55.79	90.41	73.01	54.12	
	AB23	43.58	24.14	14.52	21.11	18.64	17.91	22.88	13.98	11.11	21.83	20.09	17.29	
	AB24	104.25	110.95	46.45	51.16	50.64	65.95	87.73	59.35	67.53	62.82	67.20	36.88	
	AB25	1.14	0.62	0.18	NA	0.81	NA	0.52	NA	NA	NA	NA	0.27	
	AB26	7.70	13.40	2.28	3.26	1.19	0.27	0.67	1.21	0.86	NA	0.05	1.10	
	AB27	101.37	68.48	52.95	50.29	43.72	43.24	39.53	55.18	47.98	50.21	37.68	35.39	
	AB28	15.99	8.51	7.16	7.76	11.41	12.65	9.54	11.29	5.63	4.63	5.83	4.80	
	AB29	20.94	26.62	15.21	18.68	24.46	36.32	41.14	31.75	27.57	25.73	19.69	16.43	
	AB30	10.40	13.43	12.19	3.17	5.52	6.55	NA	NA	NA	2.49	3.36	3.53	

16.3.2 Spatial distribution of divers among sub-blocks

When considering the distribution of divers at the smallest spatial scale of sub-block within each Zone it is only possible to consider the fishery since the year 2000 when sub-block were first introduced. The variation in the number of divers is greater within the Eastern Zone than in the Western or Central Western Zone (**Figure 30**). In general, most divers fish in just a few blocks each year. In any given year a large proportion of the diving fleet (approximately 50%) will visit on average between 1- 5 blocks a year (**Figure 31**). There also appears to be some degree of site fidelity with 10% of the diving fleet visiting just one block per year (**Figure 31**).

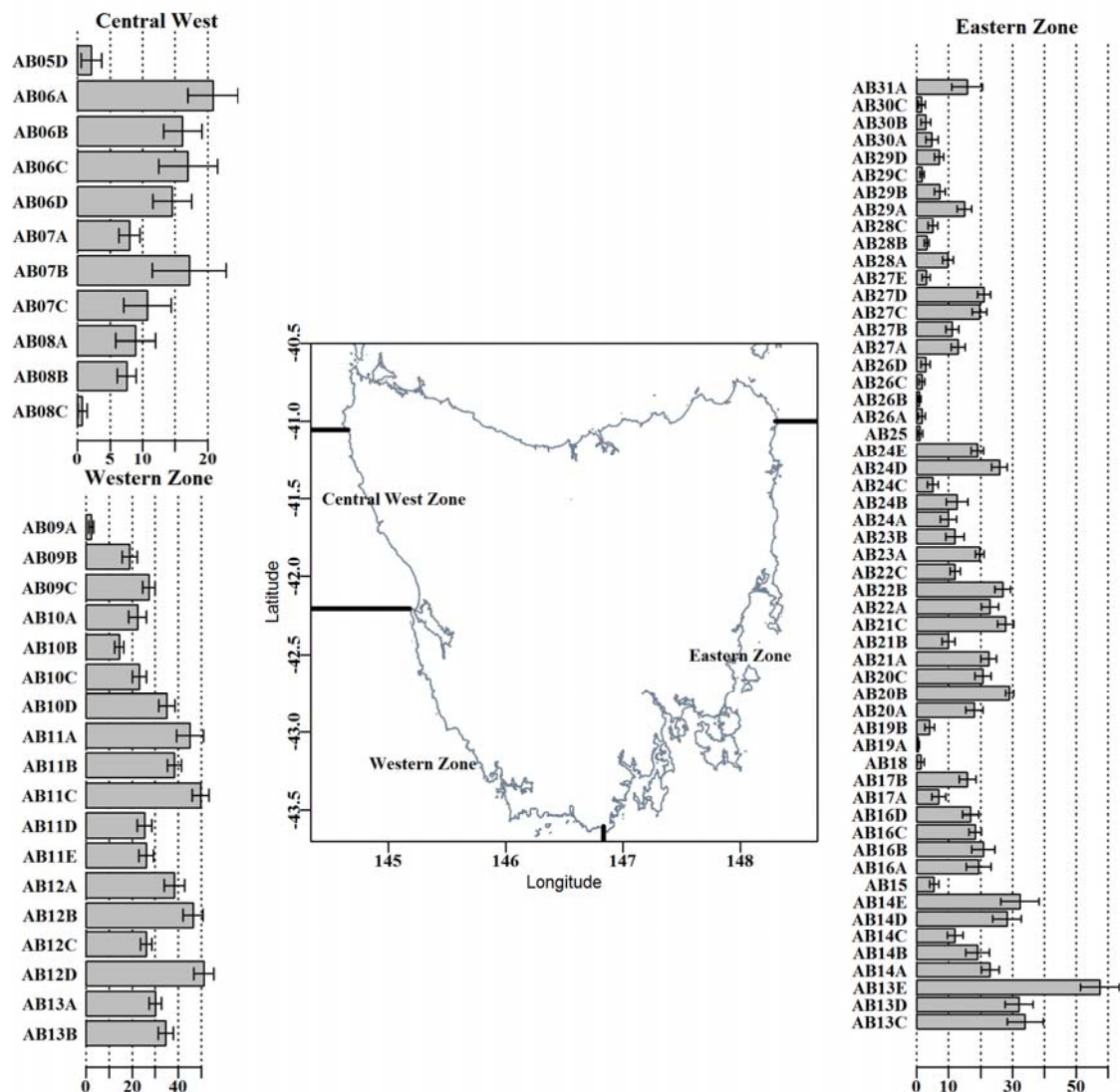


Figure 30. The average number of divers operating in each sub-block between 2000 – 2011.

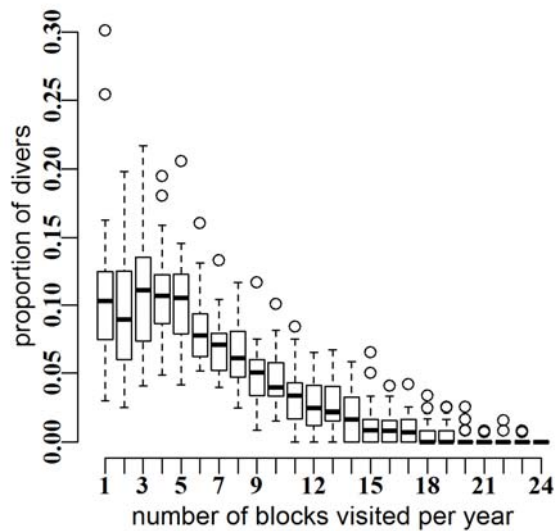


Figure 31. The number of blocks visited by each diver per year between 1985 – 2011, across the three Zones combined (the Central West, the Western, and Eastern Zone).

16.3.3 Catch-per-Diver at Different Temporal and Spatial Scales

The distribution of catches reported per diver varies between zones with divers in the Western Zone (blocks AB9 - AB12) having the highest median catch per diver (**Figure 32**). This is partly due to higher weight per abalone in the Western Zone, where abalone tend to have a larger maximum shell length (Figure 59 and Figure 63), but also because the stocks appear to be at a higher level in the west. For all zones the median catch per diver peaked around 1997 – 1999 (**Figure 32**) and during this period the TAC steadily increased from a record low 2100 Tonnes in the previous years (1990 – 1996) to 2500 Tonnes by 1999. The two major zones (the Western Zone and the Eastern Zone) both exhibited increasing catches (**Figure 35**) and relatively high CPUE (**Figure 45** and **Figure 46**).

The blocks which have well defined behaviour in terms of most predictable catch levels per diver are identified as those with the lowest variation in median catches (**Figure 32**). These blocks were mostly in the Eastern Zone (blocks 13 – 30) although generally the median daily catch for each diver there was the lowest of all the three zones considered. The greatest variations in median catches, which increased the uncertainty in this particular analysis, were found within blocks 5 – 8 in the Central Western Zone. The Westerns Zone (blocks AB9 -AB12) had the highest median catch and were the most variable so that catching behaviour was least well defined relative to the other blocks (**Figure 32**).

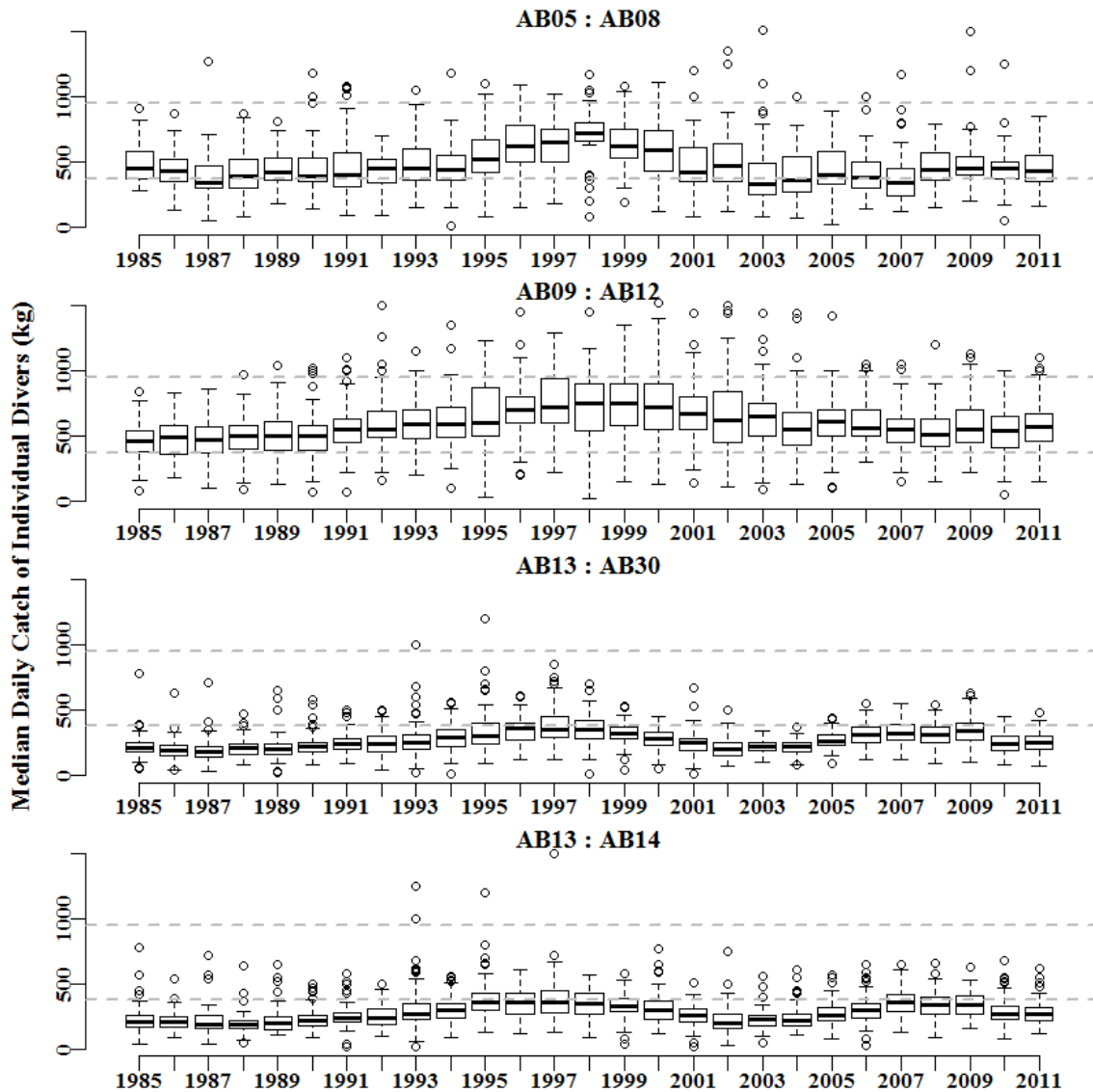


Figure 32. The median daily catch of individual divers for three zones: the Central West Zone (blocks 5-8) the Western Zone (blocks 9-12) and the Eastern Zone (blocks 13-30). Also shown is a subdivision of the Eastern Zone (blocks 13-14).

16.3.4 The dynamics of a fishery adjusted to the size of the catch

Over time the nature of the fishery in terms of available biomass changes, for example, the biomass available at the establishment of the fishery in the 1960's may not resemble the biomass available today, 50 years since the fishery formally commenced. The maximum shell length may have decreased due to fishing pressure, which early on may have favoured larger sized abalone. This section deals with the dynamics of a fishery for a given catch.

Indices estimating diver concentration, number of active divers and distribution of catch were calculated for each zone across the years 1985 – 2011 so as to illustrate any trends through time.

In the western zones, between 1994 – 1999 the catch was at its lowest but steadily increasing (Figure 35) and the CPUE was steadily improving (Figure 33 and Figure 34), During this period there were relatively few active divers in the central west and western zone (Figure 36, Figure 37) and a similar but only slight reduction in the eastern zone.

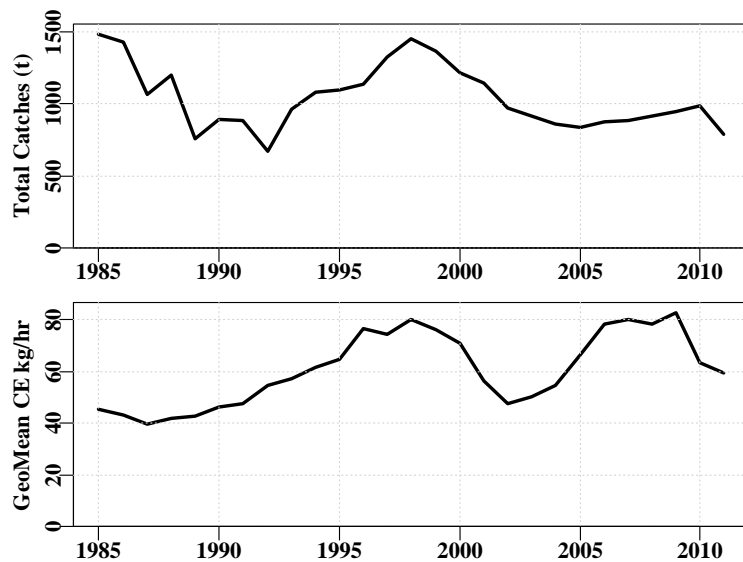


Figure 33. The total catches and geometric mean catch rates from 1985 – 2011 for blocks 13 – 31 combined.

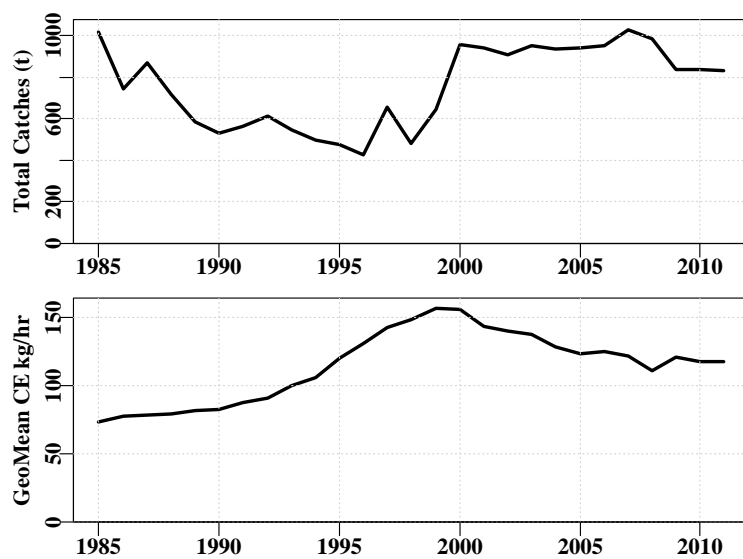


Figure 34. The total catches and geometric mean catch rates from 1985 – 2011 for blocks 9 – 12 combined.

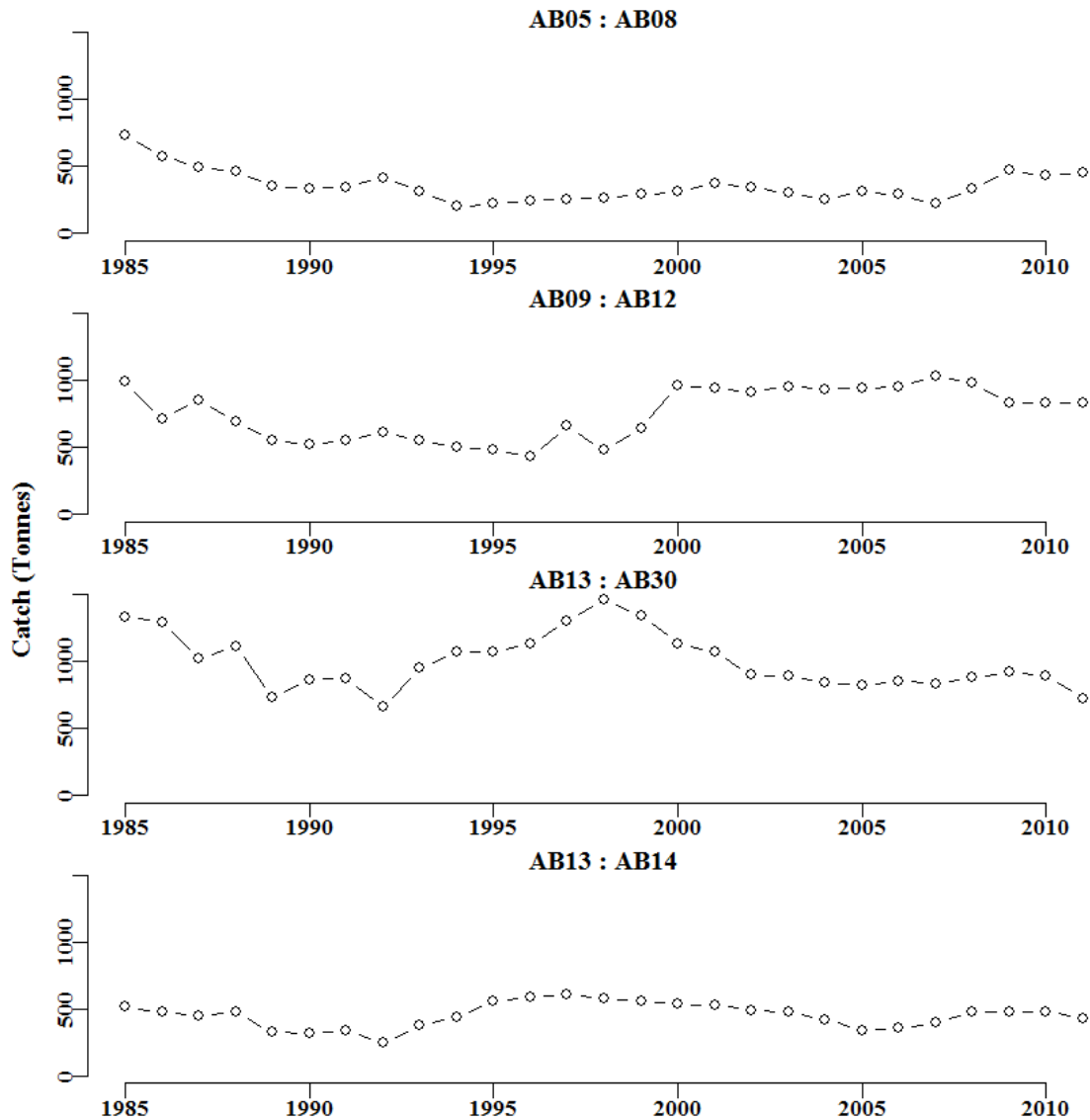


Figure 35. The total catch for three zones: the Central West Zone (blocks 5-8) the Western Zone (blocks 9-12) and the Eastern Zone (blocks 13-30). Also shown is a subdivision of the Eastern Zone (blocks 13-14).

Between 1994 – 1999 the majority of catch was being caught by relatively few divers (**Figure 38**, **Figure 39**, and **Figure 40**) and, not surprisingly, there were also fewer effective divers during this period for all zones (**Figure 41**). In general there were not many divers operating in the fishery (**Figure 41a**) and relatively few divers were active in the fishery (**Figure 41b**). It is difficult to determine if unreported team diving was taking place during this period. If so it would certainly generate misleading results and falsely indicate fewer effective divers even though this may not be the case in reality. When zonation was introduced, in the year 2000, the number of divers increased in all zones particularly in blocks 5, 6, 11 and 9. In addition, the catch was more evenly distributed among divers for all zones (**Figure 41**) and therefore each active diver became more effective. After zonation there was a notable change in the distribution of catch among divers in the Eastern Zone (**Figure 40**), which was a result of a strong reduction in the zones catches and the more even spreading of catches among the active divers.

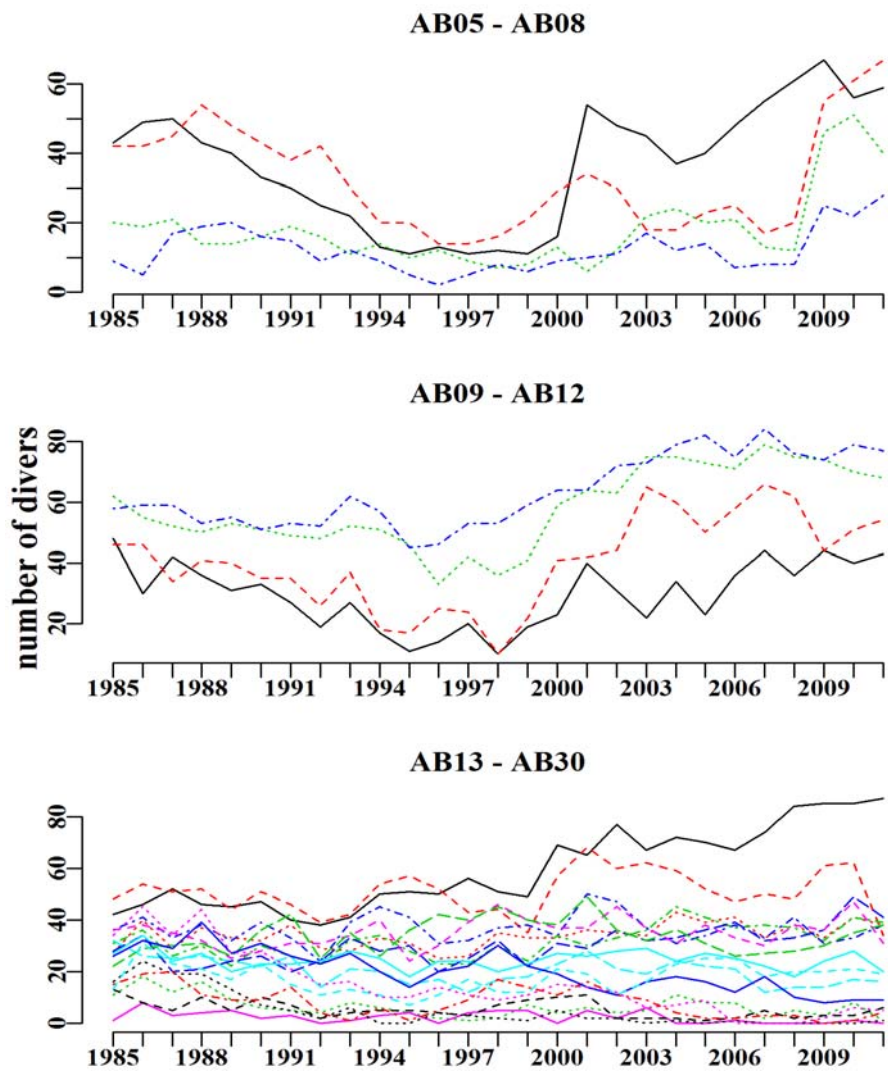


Figure 36. Number of divers per block

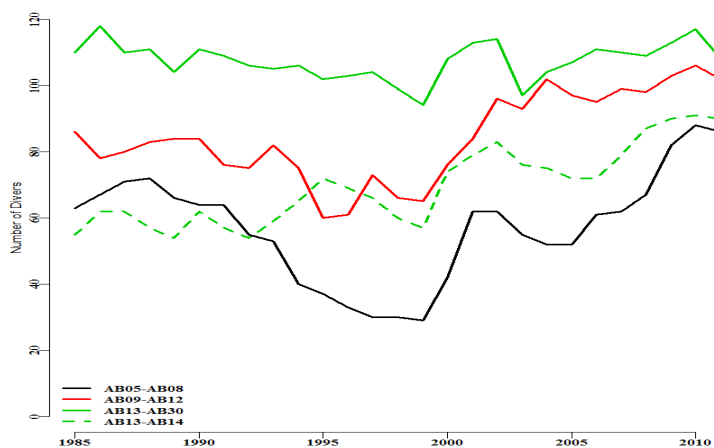


Figure 37. The total number of divers operating in each of the three zones: the Central West Zone (blocks 5-8) the Western Zone (blocks 9-12) and the eastern Zone (blocks 13-30). Also shown is a subdivision of the Eastern Zone (blocks 13-14).

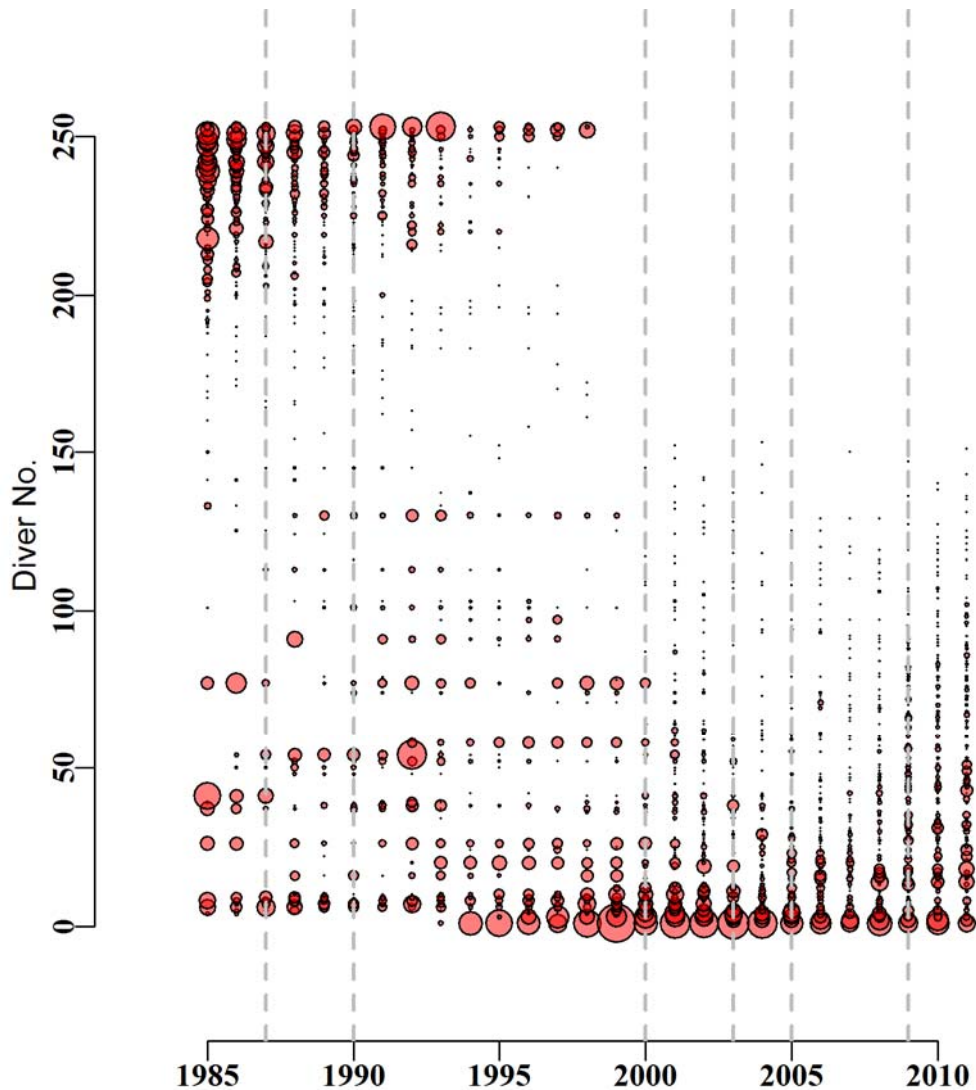


Figure 38. Central Western zone (blocks 5-8): The size of the bubble represents the size of catch for each diver for each year between 1985 - 2011, zero data on catch or effort excluded. The grey lines represent the timelines of the various management changes (refer to **Figure 21**)

16.4 Discussion

This study describes the preliminary pattern of behaviour that characterises the choice of diving locations among divers for the Tasmanian Abalone blacklip fishery. Divers exhibit behavioural responses that are typical of many other fisheries, meeting the assumptions for location choice that were anticipated a priori. Three factors affected location choice, 1) CPUE, 2) accessibility and 3) market value and there was no one single factor that predicted the choice of location. The choice was mainly determined by a combination of good accessibility and CPUE (i.e the Eastern Zone). The Eastern Zone continually attracts the majority of divers even though it has lower CPUE than Western Zone (**Figure 33** and **Figure 34**). The next single major factor (if accessibility is low) is high CPUE (i.e the western zone).

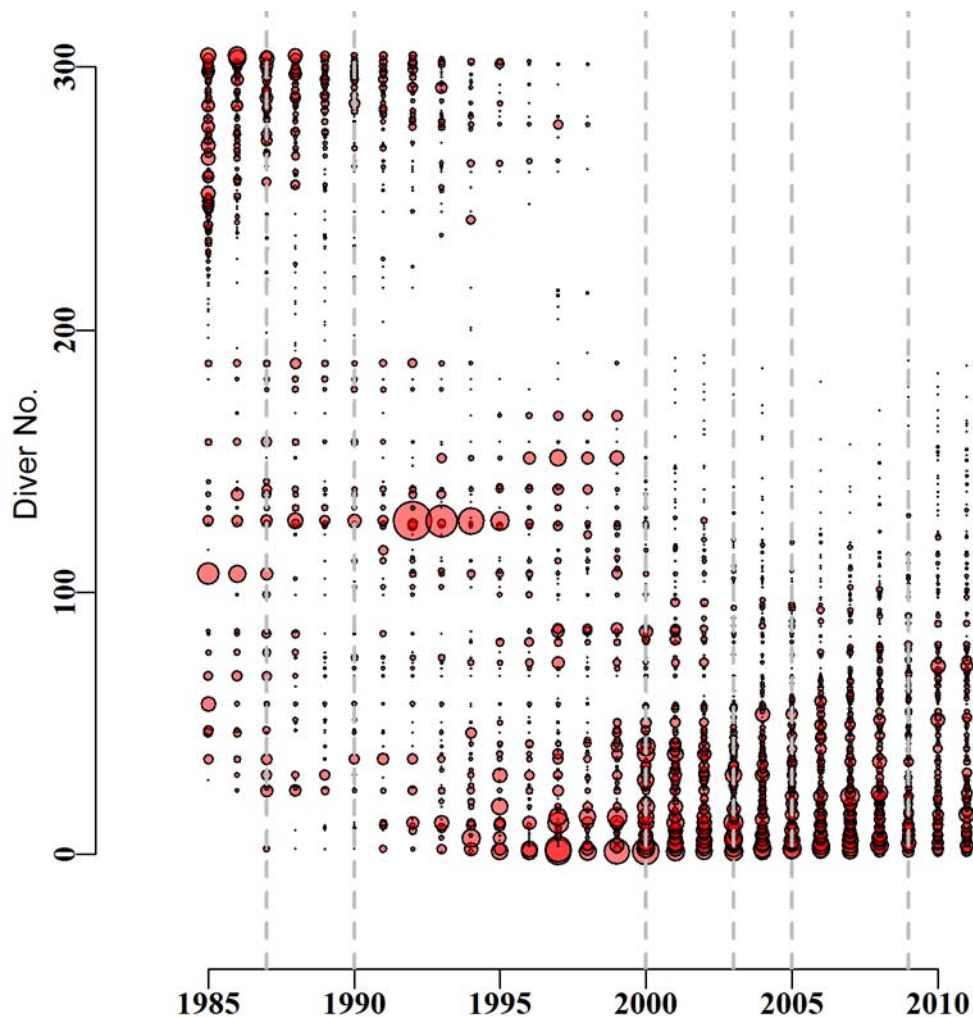


Figure 39. Western zone (blocks 9-12): The size of the bubble represents the size of catch for each diver for each year between 1985 - 2011, zero data on catch or effort excluded. The grey lines represent the timelines of the various management changes (refer to **Figure 21**)

16.4.1 Dynamics in response to reduction in total allowable catch

The effect of reducing the TAC is best observed during the 1990's when the greatest reduction in TAC occurred. The TAC was reduced from >3500 t in 1985 to 2100 t in 1989 and remained at 2100 t until 1994. The reduction seemed to affect the distribution of catch among divers leading to fewer active divers and the catch being less equally distributed among divers (**Figure 41**). During this period the median catch of an individual diver increased particularly in the Western Zone (**Figure 32**) mainly because there were fewer effective divers in the fishery (**Figure 41**). The dynamics of the fishery in the central western and western zone was less prone to change in catch per diver compared to the eastern zone (**Figure 38**, **Figure 39**, **Figure 40**). The change in the Eastern Zone was influenced by zonation in 2000 with each diver landing less catch, in absolute terms after 2000 (**Figure 38**); the spreading of effort and catch was the intention of the zonation, which was obviously successful. A slightly higher TAC from the 2000 onward meant that quota trickled out among divers and landings were more equal-

ly shared between divers in the Western Zone compared to the Eastern Zone (**Figure 41**)

The effect of reducing the TAC in the 1990s eventually led to increase in CPUE in both the Western and eastern Zone (**Figure 33, Figure 34**) and an increase in the median catch per diver for all of the major zones (**Figure 32**). Therefore a reduction in the TAC has low implementation uncertainty because it results has so far led to the intended improvements in the fishery dynamics but also has the effect clearly rewards a lower number of individual divers with higher median catch and CPUE.

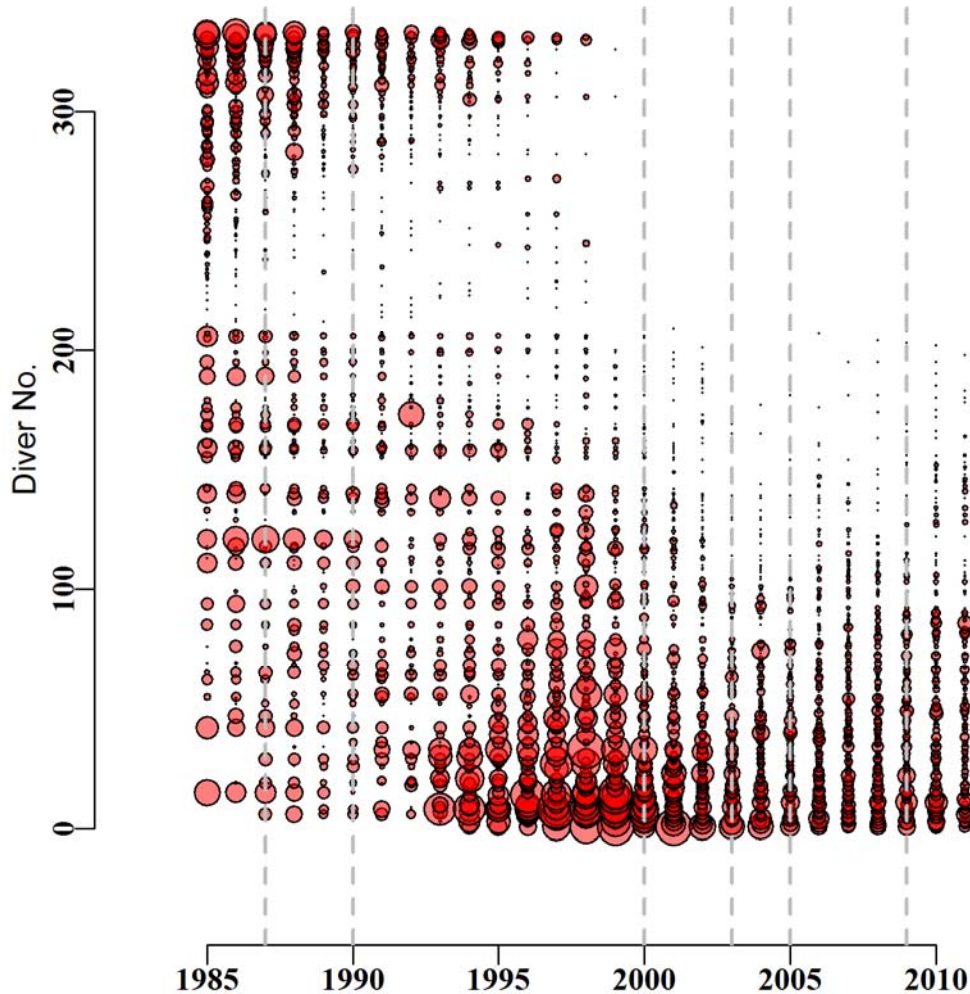


Figure 40. Eastern zone (blocks 13-30): The size of the bubble represents the size of catch for each diver for each year between 1985 - 2011, zero data on catch or effort excluded

16.4.2 Dynamics in response to changes in legal minimum length

A change in the LML did not lead to any noticeable change in the diver distribution, although it is possible that a change in the LML slightly altered the number of divers per tonne of catch (**Figure 41**). It is difficult to detect a change in the fishery dynamics possibly because the effect of the change in LML was very slight (+- 4mm) . The

greatest change in LML was 5mm and this occurred between 1986 – 1987. With reliable data only commencing in 1985 it is difficult to discern any trends pre 1986 to post 1986.

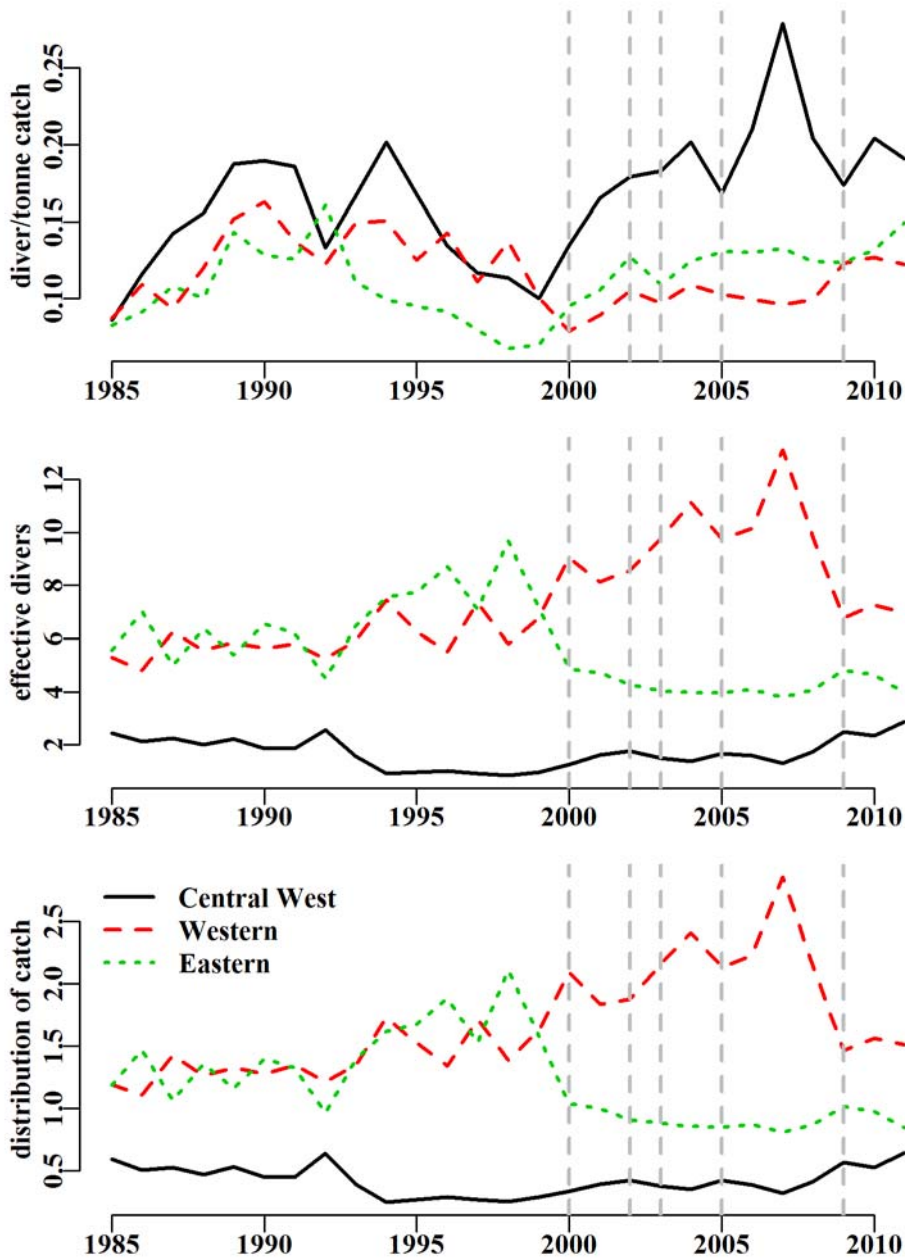


Figure 41 Characteristic between the distribution of catch and the number of divers operating in the Tasmanian blacklip fishery, zero data on catch or effort excluded. The grey lines represent the timelines of the various management changes after the year 2000 (refer to **Figure 21**)

16.4.3 Dynamics in response to spatially designed management controls

Persistent long-term habits and diver experience are evident in the predictability of the number of blocks they are likely to visit when fishing (**Figure 31**) however, management regulation such as zonation can alter the choice in fishing location in a predictable manner. For example it was evident that although the majority of divers visit up to 1- 5 blocks per year (**Figure 31**) the introduction of zonation resulted in a reallocation of effort as intended, with the total number of divers increasing on the Western and Central Western Zone (**Figure 36, Figure 37**). The aim of establishing zonation between the eastern and Westerns Zone in 2000 was to redistribute effort and catch away from the Eastern Zone and towards the Western zone. The management induced zonation produced the intended outcome with more divers and higher catches from the Western Zone after the year 2000 and a decrease for the Eastern Zone (**Table 15**). This clearly indicates that zonation along with catch caps had low implementation uncertainty as it resulted in the intended outcome.

16.4.4 Area specialization and habits

As with other fisheries the choice of selecting a distinct fishing location depends on the perceived utility of the fishing area (Tidd et al., 2012). The utility can be characterised by previous economic success. In addition the spatial distribution of divers in the abalone fishing industry is also determined by past habitual behaviour or previous experience and fuel price.

Cost in terms of fuel and transport costs and proximity to port of landing is an important driver of location choice (Tidd et al. 2012). The Value per Unit Catch (VPUE) is a measure of CPUE against the costs of obtaining those catches (Tidd et al. 2012). In the abalone fishery the potential revenue can be determined by the amount of quota allocated to divers. Quota that is metered out to divers in small quantities will result in lower income to the diver in absolute terms. Therefore with lower income, the diver decision is to opt for more accessible areas that have lower fuel and transport costs thereby maximising the VPUE. If individual divers receive a large quota then this may enable divers to fish in more inaccessible or deeper area with good yields because the costs in transport may be met by the large return from the larger quota and therefore the VPUE may increase.

Experience has a bearing on location choice (Tidd et al. 2012). Abalone divers rely on previous experience of where the relatively inaccessible but high yielding areas are likely to be, and therefore diver experience is another factor that may determine spatial distribution in the abalone fishery

The choice of fishing location can vary between divers and this may be caused by individual variation in experience and other unexplained factors (Tidd et al. 2012). For example some heterogeneity in choice of fishing location is exhibited by individual divers with few divers selecting multiple blocks while other divers exhibit area specialisation (**Figure 31**).

Fishery managers aim to develop strategies to sustain a fishery both ecologically and also profitably in the face of increasing fuel costs, varying stock levels, changing regulations, and market conditions (Abernethy et al. 2010; Tidd et al. 2012).

An understanding of the processes that influence the spatial dynamics of the fishery (location choice and distribution of catches and CPUE) is a step toward reducing implementation uncertainty so that the intended effects of management may be realised. These processes are invariably complex and interrelated. This study provides the information required for an initial understanding of diver choice in fishing location both spatially and temporally in response to management controls.

The field of Fisheries Science is increasingly becoming interdisciplinary and is not solely based on biological assessments (recruitment, spawning-stock biomass). The field encompasses economic (fuel market prices), social (employment), and regulatory objectives (quotas, LML's, limited entry) (Tidd et al. 2012). The benefit of the interdisciplinary approach is that fishers and management can optimise their fishing strategy from catch information rather than fish under uncertainty to minimise risks. Minimizing implementation uncertainty may improve the management of the fishery (Fulton et al. 2011) especially when considering seasonal closures and marine protected areas, MPAs (Tidd et al. 2012).

One approach that offers the prospect of addressing implementation uncertainty is to use Management Strategy Evaluation (MSE) framework and simulate alternative management options and responses in fleet dynamics (Fulton et al. 2011; Tidd et al. 2012)). Knowledge of scale-fish fisheries is relatively well documented, however, more work similar to that reported here is required for abalone fisheries if the range of behaviours in response to management changes is to become better understood.

17 Empirical Performance Measures

17.1 Introduction

Performance measures (PMs) can relate to numerous variables concerning the fishery and the stock being fished. There are two major forms of PM, those termed empirical and those termed analytical. Where a PM is based directly on data collected from the fishery these would be empirical PMs and where such data is further analyzed, perhaps derived from a formal model, and the derived statistics used as the PMs, these would be analytical PMs. Empirical PMs include such things as catch-per-unit-effort (CPUE) or the proportional distribution of catches across different areas within a zone, while analytical PMs include model derived statistics such as stock spawning biomass or fishing mortality. Many more assumptions are required when using analytical PMs, however, because many of these are taken to relate directly to the fished stock's dynamics through time they are often afforded a higher value in the interpretation of a stock's status. Which type of PM is used in a particular situation is directly a function of what harvest control rules (HCRs) are in place and, more fundamentally, towards what objectives the given fishery is being managed. If the HCR adopted relates directly to spawning biomass then empirical PMs are not capable of addressing such requirements, although they can be used if treated as acceptable proxies for stock biomass (Haddon, 2012).

If multiple empirical PMs are used in combination (for example trends in catch rates and trends in the commercial length frequencies) this can be more informative than single empirical PMs. Using empirical PMs, and possibly multiple PMS, is an option when there is insufficient data available to fit a fully operational integrated analysis; in effect the integration of such disparate data streams is being done qualitatively rather than formally and quantitatively.

17.2 Catches

Catches by themselves provide little information regarding their sustainability through time (Hilborn and Branch, 2013), although if a long time series is available and show no sign of diminishing this would constitute at least circumstantial evidence that an area can be productive at the observed levels. However, having said that, in section 20.4.5, when considering the changes in the numbers of abalone required to catch a TAC, long term simulations of fishing a zone demonstrated that some zones can be productive over a 50 year period while depleting slowly. This is possibly a reflection of the flatness of the productivity or yield curve for each zone (**Figure 83**) implying that catches greater than the maximum yield can be sustained for quite a long time as spawning biomass declines slowly. Thus, catches by themselves are not an adequate PM of a stock's status or productivity. However, if combined with such things as observed length frequencies in the catch or with catch rates, then these together can indicate whether a particular catch level is leading to declines in the resident stock.

Using the geographical distribution of catches is adding a different kind of information to simple catch data. Again in combination such data should be capable of indicating whether a contraction in productive areas is occurring or perhaps serial depletion could

be occurring. Such combined information should certainly be directly useful in preventing continued over-exploitation of particular areas. For example, where a TAC expected to be taken from a large area was continually focused in a much smaller area. Whether catches can provide information on the expected productivity of an area has still to be determined.

17.3 Effort

Like catch, the effort applied through time within a zone only allows very crude comparisons between years, although this may have direct interest when considering economic inputs and possible related PMs. Fishing effort needs to be combined with other data streams to become informative about the stock status. Those other data streams can include catch, location, and time of year. When combined with any of these things (or other factors) then effort can become very informative concerning the dynamics of a given fishery and possibly of the stock being fished.

Defining effort is another issue when it involves a diver fishery. Ideally effort would be characterized as the time spent underwater searching for abalone. This ignores the time it takes to travel to a location, which may have importance when divers decide where to fish; but that deals with fishing decisions rather than stock abundance. However, at what level the available information should be integrated has never been considered in detail. Should effort be defined in hours per dive, or should it be hours per day, or what level of summary should be used to best separate noise from information. There is a suggestion to use as a PM the number of short dives where the diver decides it is not worth while fishing in a spot despite prior expectation. This will be explored once the GPS data logger data builds up a time series across years. The mixture of searching and fishing complicates the interpretation of effort and catch rates. With the advent of depth data loggers and the complete coverage of the Tasmanian fishery the opportunity to explore this question is becoming available, but insufficient data have been collected to date to work on that here.

17.4 Catch-Per-Unit-Effort (CPUE)

17.4.1 Introduction

An index of relative abundance is commonly used in the monitoring of wild populations and when assessing the status of any fished stock. In fisheries, estimates of catch per unit effort (CPUE) are often used for this purpose (Hilborn and Walters, 1992; Punt *et al.*, 2001; Little *et al.* 2011), however, the validity of this use depends on changes in CPUE being proportional in some way to changes in abundance. The underlying assumption behind this use of CPUE is that there is a simple relationship between the estimated catch rate and the amount of exploitable biomass available each year. This is usually expressed as:

$$\frac{C}{E} = qB^\lambda e^\epsilon \quad (5)$$

where C is catch, E is effort, q is the catchability coefficient, literally the proportion of the exploitable biomass, B , expected to be taken with one unit of effort, λ is an exponent that can be used to describe non-linearity in the relationship between CPUE and ex-

exploitable biomass, although generally λ is assumed to equal 1.0, and the e^ϵ represents the log-normal errors usually assumed for catch rate data.

Unfortunately, there are many circumstances where CPUE is not linearly proportional to abundance. Harley *et al* (2001) estimated the λ values for over 200 datasets where CPUE estimates could be compared with fishery independent abundance surveys. The majority of the scalefish fisheries they considered expressed λ values between 0.64 – 0.75, which implies that CPUE does not decline at the same rate as the stock abundance. If such hyper-stability is unknowingly present this could bias any management advice based on using CPUE as an abundance index. Hyper-stability can arise where catch rates exhibited by a group of fishers can be modified by the fishers changing their behaviour or fishing patterns. Classic examples of this can be found in fisheries involving hand collection, such as with abalone species, where divers collect the animals individually from the sea bed. In some circumstances divers can change their behaviour in order to maintain catch rates despite a decline in abalone availability, thus CPUE becomes hyper-stable and hence less informative about abundance. In a formal description of a full size-based assessment model (Breen *et al.*, 2003) estimates of the λ parameter for an abalone (*Haliotis iris*) on the south coast of the South Island in New Zealand generated estimates between 0.96 – 1.16, and they concluded that the data available had no information about the asymmetry value and so they generally used a value of 1.0. However, an assessment of stocks within PAU 07 (Breen & Kim, 2005), which includes the north coast of the South Island, estimates of λ ranged between 0.62 – 0.64.

The paradigm of abalone catch rates being uninformative was emphasized by Sloan and Breen (1988) who summarized reviews by Breen (1980) and by Federenko and Sprout (1982). Their main point was that abalone fisheries are prone to serial depletion and this led to catch rates becoming uninformative about relative abundance. If fishers sequentially harvest and deplete separate beds then catches can be maintained or even increased with no comparable decline in catch rates becoming apparent despite the stock as a whole declining; this would constitute the classic cause of hyper-stable catch rates.

More recently, the risk of serial depletion unknowingly occurring has been identified as relating to the geographical scale of catch and effort reporting being generally much greater than the geographical scale of the fishing operations (Prince, 2005). The mismatch between the scale of reporting and the scale of the fishing is what makes the catch rate data vulnerable to the occurrence of hyper-stability. However, such hyper-stable catch rates stem from fishers behaving in a particular manner, thus hyper-stable catch rates are merely a potential outcome and are not a necessary outcome. If, for example, a fishery was very well known and all sites were visited every year, and there are workable management strategies in place (such as a TAC, an LML, and options for ensuring a spreading of effort), then the risk of serially depleting a region would be reduced. Such serial depletion would be mainly a risk at the start of a fishery before it had been fully fished or had management capable of conserving sustainable fishing being introduced. Again the details of such behaviour will become available for further investigation once a time series of GPS data logger data is built up.

There remain important abalone fisheries in New Zealand and Australia with, for example, about 30% of all remaining wild caught abalone in the world market being taken in

Tasmanian waters (FAO, 2006). Despite the paradigm that commercial catch rates are unreliable and uninformative (Prince and Hilborn, 1998), all formal abalone stock assessments in Australia and New Zealand include the option of using commercial catch rates as an index of relative abundance, and often rely greatly on such time series (Breen and Smith, 2008; Breen *et al.*, 2003; Gorfine *et al.*, 2005; McKenzie and Smith, 2009; Worthington *et al.*, 1998). Less formal, empirically based fishery assessments in Tasmania and South Australia also use abalone catch rates to inform management (Burch *et al.*, 2011).

The management of abalone fisheries has had varied success around the world, with a number of important fisheries collapsing (Hobday *et al.*, 2001). The lack of success has been at least partly attributed to the inability of abalone catch rates to provide a useful measure of relative abundance and thereby failing to detect severe declines in stock abundance until too late to prevent collapse. If catch rates are related to stock size in a mature fishery then it would be expected that if high catches were maintained or increased then catch rates would correspondingly decline and, conversely, if catches declined then catch rates would tend to increase. While a lag between changes in catches and corresponding changes in catch rates may be expected, nevertheless the directions of any change should be consistent. We will thus examine the history of the abalone fishery around Tasmania and determine whether or not the observed changes in catch rates are related to changes in the catch taken from the different areas. This will help us answer the question: Are there circumstances where hyper-stability does not occur in abalone fisheries so that CPUE can become informative about relative abundance?

As well as identifying hyper-stable catch rates as a problem for abalone fisheries, Sloan and Breen (1988) also pointed out that abalone catch rates are made variable by divers having different reporting practices for effort (e.g. effort as hours on the water or sometimes hours underwater). This source of variation remains a problem but many factors other than the stock abundance can also influence the apparent CPUE (for example, the diver doing the fishing, the month of fishing, the location of fishing, etc). The standard approach used to account for the effects of such factors is to apply statistical standardization to the CPUE data. While this ought to be a significant improvement on using the raw catch rates, it is the case that standardizations that use diver as a factor would fail to account for the situation where some of the divers altered their usual behaviour in the face of decreased availability of their target species.

Given that diver behaviour can be very influential, the impact of the diver doing the fishing would appear to be very important to observed catch rate trends; and this is in fact what is found in all abalone catch rate standardizations in Tasmania. The intuition that catch rates of the top performing divers are biased low by the catch rates of the other divers, especially in periods of low stock availability, is so strong that the Tasmanian Abalone Industry Council requested that an analysis of the catch rates of the top 30 divers (in terms of total catch across a ten year period) be used as the foundation for management decisions rather than using an analysis based on all divers. To test the reality of this intuition, separate standardizations of the catch rate data from the top 30 divers (in terms of total catches and consistently high catch rates) and for all divers were conducted on data from multiple regions around the Tasmanian fishery.

Abalone populations are also notorious for the variation in biological properties exhibited by populations separated by even relatively small distances. Comparing how the performance of different sets of divers varied in different regions also permitted an examination of how catch rates varied between regions within the same quota zones.

An aim of this present work is to re-examine the assumption that abalone catch rates are uninformative about stock status. This will involve a consideration of the occurrence of hyper-stability and of the main factors influencing CPUE. If CPUE is to continue to be used in discussions of stock status then the validity of its use needs to be examined in case any management advice stemming from such assessments is being biased or distorted.

17.4.2 Objectives

1. To identify to dominate factors that affect catch rates, as identified in statistical standardizations, and determine whether these differ between coasts.
2. To determine the impact of different areas along a coast, or within a zone, as a factor in catch rate standardizations by comparing the catch rates of different regions within single quota zones.
3. To test whether abalone CPUE in Tasmania appears to be hyper-stable by determining whether CPUE responds in complementary ways to changes in catch levels through time (as catches go up CPUE should come down, and visa-versa).
4. To determine the impact of diver as a factor in catch rate standardizations by comparing the catch rates of the top 30 producing abalone divers when analysed separately from the rest of the divers with those produced when all divers are analysed together.

17.5 Methods

17.5.1 The Data

The Tasmanian Department of Primary Industries, Parks, Water, and Environment (DPIPWE) are tasked with gathering and collating the information contained in each of the daily docketts produced by each diver detailing the catch and effort of abalone around the State. Since 1985 this data has been provided at the scale of the abalone statistical reporting blocks (e.g. Tarbath and Gardner, 2011), and since 2000, each reporting block has been subdivided into an array of subblocks; these are the finest scale of reporting currently available; although now that GPS data loggers are required in the Tasmanian fishery, as the data continues to be collected the scale of data collection will become at the same scale as the operation of the fishery.

Staff at the Marine Laboratory of the University of Tasmania first check the data for errors and conduct range checks on input data fields. It is this checked data that is used in these analyses. Further to those checks, in order to use diver name as a factor in the standardizations the names have to be made usable. This entails stripping out all punctuation and spaces because these have not been entered consistently through time. Thus a hypothetical diver “A. Diver” can be found in the database as “A Diver”, “A. Diver”,

“A.Diver”, and A., Diver”, plus other combinations of spaces and punctuation; for these to be recognized as the same diver by the software in the analysis it is necessary to remove the spaces and punctuation.

17.5.2 Statistical Standardization

The idea behind statistical standardization of catch rates is that the relationship between the exploitable biomass and catch rates is potentially obfuscated by other factors which can influence catch rates independently of exploitable biomass. By standardizing for the various factors influencing the fishery, emphasis can be placed upon the year factor (or some other time-period) which ought to more closely reflect stock biomass after the analysis.

The four main factors available for use in the analyses here include the Year of catch, the Month of catch, the Diver doing the catching, and the statistical Block in which fishing occurred (**Table 14**). In addition, there may be significant interactions between Block and Month, or between Diver and Block, or possibly between Diver and Month. Of course if the focus is on a single block then a number of these factors become redundant. Average CPUE varies between divers and there is a turnover through time among the active divers. Changes in seasonality and the inter-annual distribution of effort among blocks also contribute to variation in CPUE (and the Month and block factors attempts to account for that). The standardisation required daily records from individual divers and the CPUE data was normalised by using a log-transformation. A General Linear Model was used with this log-transformed data rather than using a Generalized Linear Model on the untransformed data with a log-link; this has advantages in terms of normalizing the data while stabilizing the variance, which the Generalized Linear Model approach does not always achieve appropriately (Venables & Dichmont, 2004). This relatively simple analytical approach means that the exact same methods can be applied to all areas in a relatively robust manner. The statistical models were variants on the form: $\text{LnCE} = \text{Year} + \text{Diver} + \text{Month} + \text{Block}$. In addition, there were interaction terms which could sometimes be fitted, such as Month:Block or even Month:Diver; because there are so many different divers through time this latter interaction term requires a very large amount of data and invariably failed to produce useful or significant results. The CPUE, was thus statistically modelled with a normal GLM on log-transformed CPUE data:

$$\text{Ln}(CPUE_i) = \alpha_0 + \alpha_1 x_{i,1} + \alpha_2 x_{i,2} + \sum_{j=3}^N \alpha_j x_{ij} + \varepsilon_i \quad (6)$$

where $\text{Ln}(CPUE_i)$ is the natural logarithm of the CPUE (kg/h) for the i -th record, x_{ij} are the values of the explanatory variables or factors j for the i -th record and the α_j are the coefficients for the N factors j to be estimated (α_0 is the intercept, α_1 is the coefficient for the first factor, etc.

Up to five different log-linear models were fitted and compared in an effort to account for the effects of Year, Diver, Month of fishing, statistical block, and any interactions between Block and Month (**Table 14**). Interaction terms involving divers were insignificant, but this may have been due to the limited amount of data from many of the divers in any single year. All factors were treated as categorical. The optimum statistical mod-

el was selected on the basis of the Akaike's Information Criterion and after assessing the improvements to the adjusted R^2 after fitting each model.

Table 14. Statistical models applied to the log-transformed abalone catch rate data. Models 4-5, which include Block, cannot obviously not be applied to single blocks.

Model 1	LnCE ~ Year
Model 2	LnCE ~ Year + Diver
Model 3	LnCE ~ Year + Diver + Month
Model 4	LnCE ~ Year + Diver + Month + Block
Model 5	LnCE ~ Year + Diver + Month + Block + Month : Block

17.5.3 The Overall Year Effect

For the lognormal model the expected back-transformed year effect involves a bias-correction to account for the log-normality; this ensures the estimates relate to the mean of the distribution rather than the median (Hastings and Peacock, 1975):

$$CPUE_t = e^{(\gamma_t + \sigma_t^2/2)} \quad (7)$$

where γ_t is the Year coefficient for year t and σ_t is the standard deviation of the log transformed data (obtained from the analysis). In this case where the mean is being estimated the bias adjustment is positive, when it is being simulated it needs to be negative, see equation (71). To remove spurious visual effects influencing comparisons between trends brought about by different scales between areas, the year coefficients in each series were all divided by the average of the year coefficients for each series to put them all on the same scale:

$$CE_t = \frac{CPUE_t}{([\sum CPUE_t]/n)} \quad (8)$$

where the $CPUE_t$ are the yearly coefficients from the standardization, $[\sum CPUE_t]/n$ is the arithmetic average of the yearly coefficients, n is the number of years of observations, and CE_t is the final time series of yearly index of relative abundance. If the original scale of kg/hr is required, then an approximation could be generated by multiplying these parameters by the simple geometric mean of nominal catch rates estimates across all years considered.

17.5.4 Information Content

If catch rates are informative about stock sizes and their dynamics in response to fishing then the expectation is that rising catches will lead to declines in catch rates and declining catches should enable catch rates to increase. This pattern appears to occur on both the east and the west of Tasmania, where reductions in catches were followed by increases in catch rates and increased catches also appear to have led to decreases in catch rates. If the time lag between these events is consistent through time, this would suggest both that there is a link between catches and subsequent catch rates and therefore that catch rates are reflecting the dynamics of the fished stock.

To test this relationship, linear regressions between catch rates and catches were carried out with sequentially increasing time lags by which the catch rates were pushed backwards. If there is a relationship between the two this relationship would be expected to be negative. This would imply that catches now should influence catch rates in the future, with high catches reducing future catch rates and relatively low catches allowing future catch rates to increase. By comparing the resulting correlations and the statistical significance of each relationship the optimal time lag can be determined.

17.5.5 Western Zone: Blocks 9 – 12

In the south west of Tasmania are four statistical reporting blocks (9 – 12) (Figure 42).

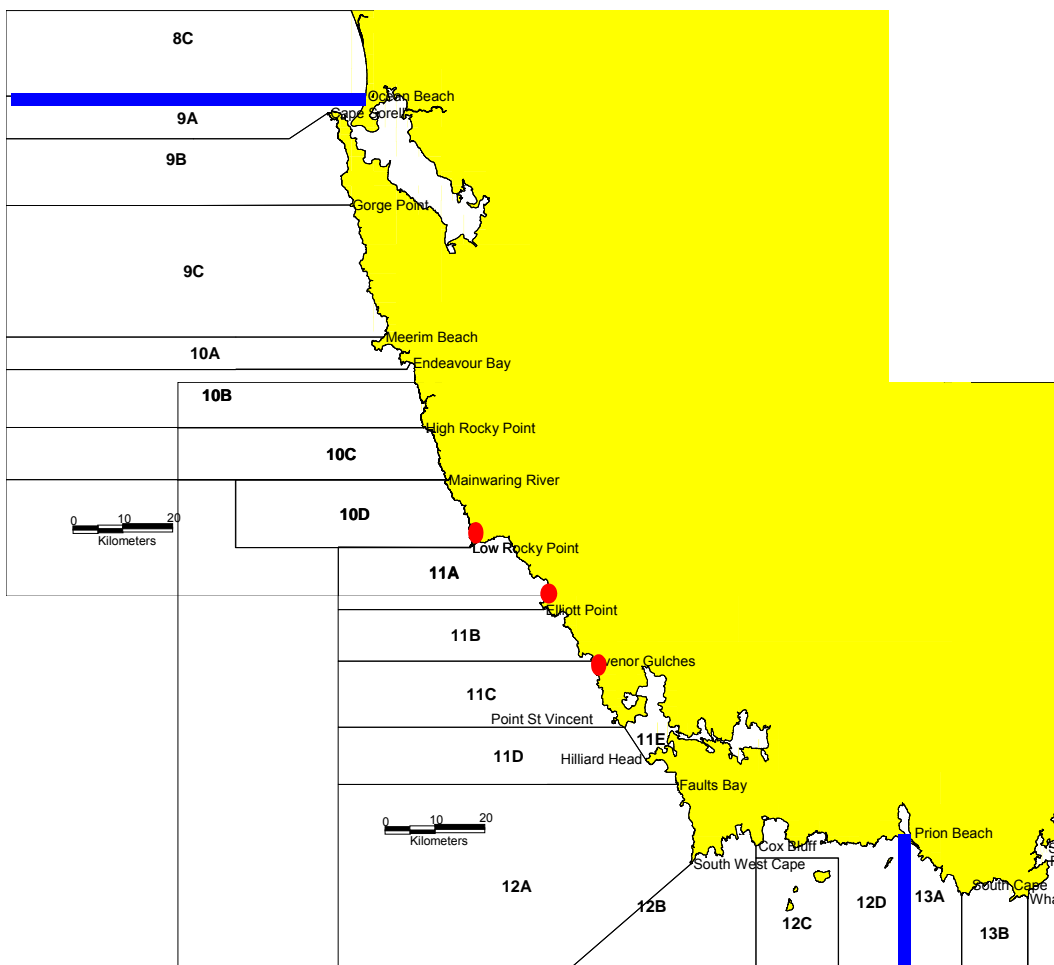


Figure 42. Schematic map of the south west of Tasmania with the statistical reporting blocks and sub-blocks for abalone (copied with permission from Tarbath and Gardner, 2011). Blocks 9, 10, 11, and 12, form complete reporting areas and are bracketed by the heavy blue lines. The three red dots identify the locations for which growth information, from tagging, is available (Black Island, Giblin River, and Hobbs Island respectively; see Haddon *et al.*, 2008). The new Western zone (now reverted) was comprised of blocks 9, 10, 11, 12, and subblocks 13A and 13B (from Ocean Beach to Whale Head). This standardization is restricted to blocks 9 to 12 so as to extend the time series back to 1985.

Together with subblocks 13A and 13B they constitute almost a 900 t fishery for abalone (**Figure 45**). Subblocks have only been defined since the year 2000 and so only blocks 9 – 12 are considered in order to have the time series extend back to 1985. A limit of 1985 was chosen because at that time quotas had been introduced, a new log book system had been introduced and the data quality improved dramatically.

17.5.6 Eastern Zone: Blocks 13 – 30

The eastern zone is defined as including subblocks 13C up to subblock 31A and the details of these divisions were produced when subblocks were introduced in 2000. For this present analysis subblocks are ignored and the east coast is defined as blocks 13 – 31 (**Figure 43**).

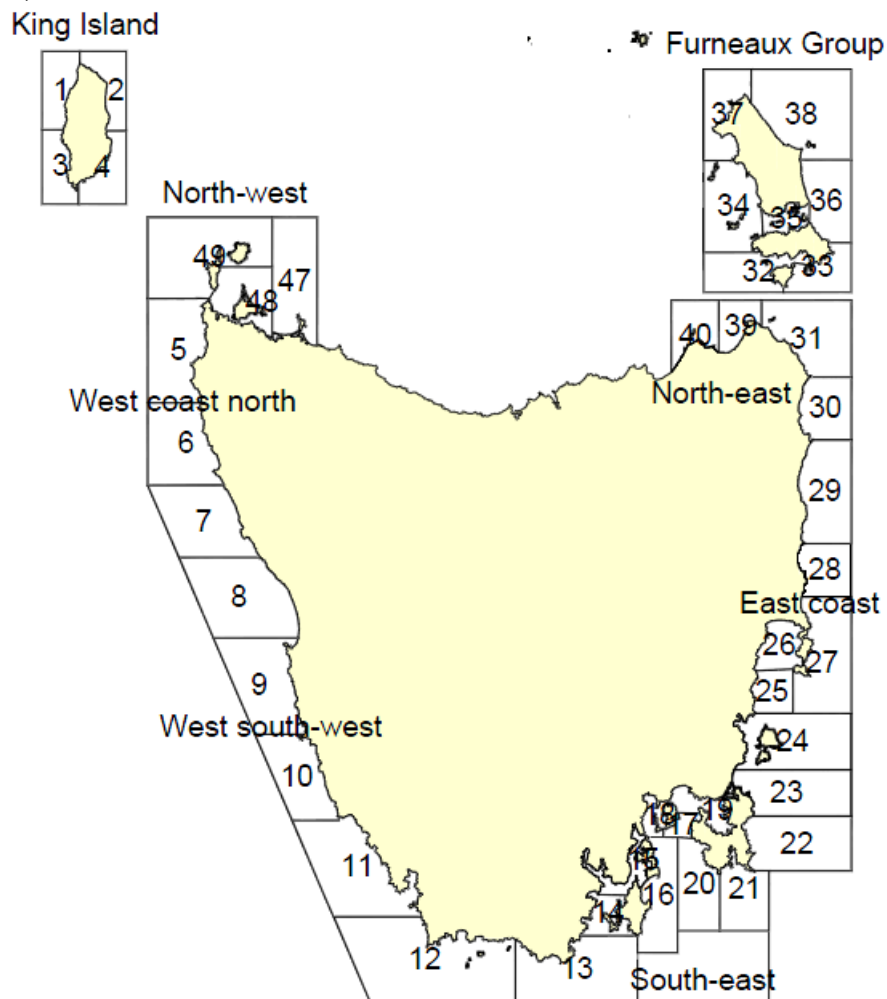


Figure 43. Schematic map of Tasmania illustrating the eastern zone, which stretched from half way through block 13 in the south to half way through block 31 in the north (copied with permission from Officer and Tarbath, 2000).

The eastern zone in Tasmania includes the Actaeon Island region which is a remarkably productive region that occurs in a small part of subblock 13E (**Figure 44**). The 18 separate blocks along the east coast include some which have relatively high catches and others with almost no catch. For example, Blocks 15, 19, and 26 are omitted because for

many years catches in these blocks are 3 tonnes or less, although occasional years of somewhat larger catches have occurred; across the years 1985 – 2011 the catches in the omitted three blocks constitute 1.18% of all catches. These blocks were omitted because the catch rates in those regions would not be expected to represent the rest of the stock. In this case the east coast is subdivided into four regions, blocks 13 + 14, blocks 16 – 21, blocks 22 – 24, and blocks 25 – 30.

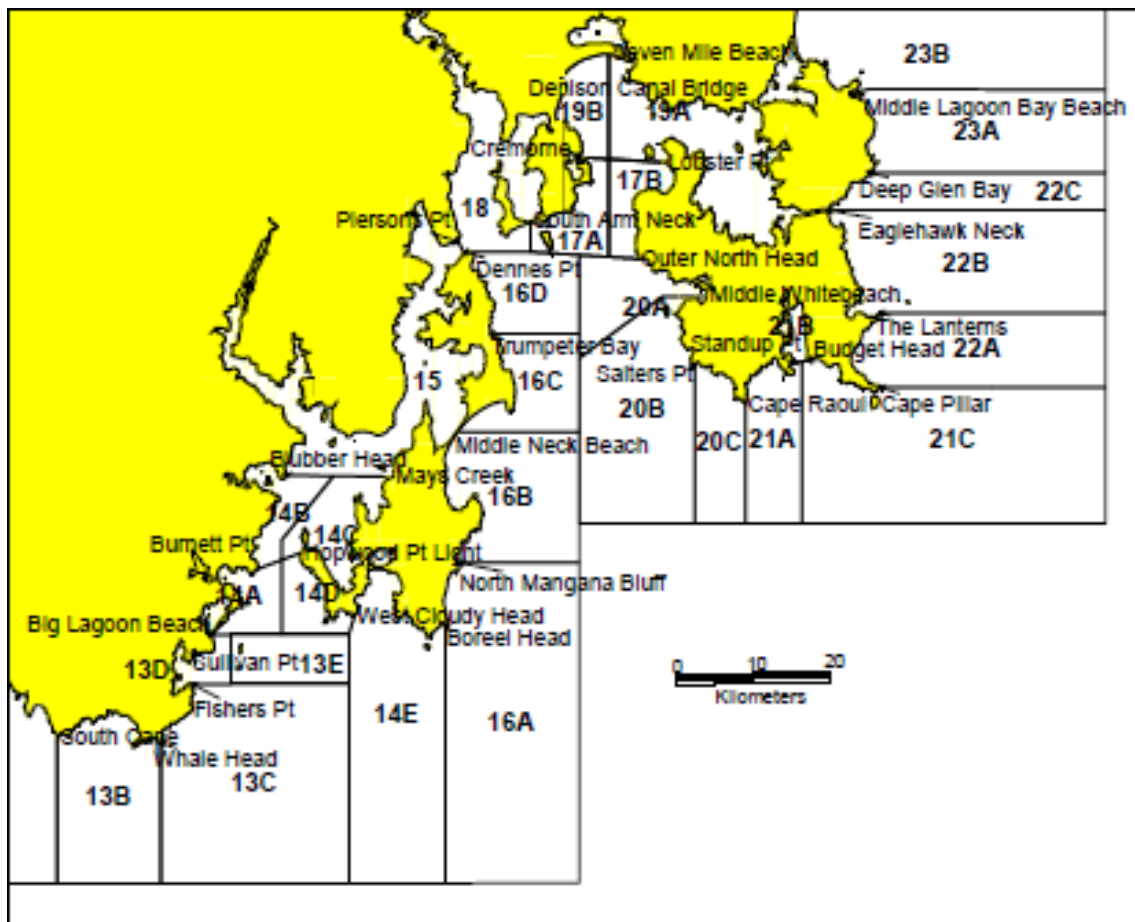


Figure 44. Schematic map of Tasmania illustrating the south eastern zone in Tasmania (copied with permission from Tarbath and Gardner, 2011). The Actaeons Islands are at the western end of subblock 13E. The Eastern zone begins in 13C, while 13A and 13B are now in the western zone.

17.6 Results

17.6.1 Summary Statistics

The western blocks featured make up half the west coast while the eastern blocks constitute the whole east coast. Catch rates are generally higher on the west coast than the east but the weather conditions in the west are often difficult and fishing tends to be episodic along with the fine sea conditions.

The west coast has exhibited a long rise followed by a slow fall since 2000, while the east coast has exhibited two oscillations up and down in terms of catch rates.

Table 15. Total reported catches and geometric mean catch rates for the statistical reporting blocks considered on the west coast (9 – 12) and the east coast (13 – 31). Catches are in tonnes, catch rates are in kg/hr, and Records are the number of reported dives.

Year	Western Blocks 9 - 12				Eastern Blocks 13 - 31			
	Catch	CPUE	Records	Hours	Catch	CPUE	Records	Hours
1985	1018.884	73.565	2116	12408	1482.402	45.378	6731	29977
1986	742.347	78.223	1477	8540	1427.356	43.388	6911	30137
1987	868.023	78.311	1721	10076	1066.937	39.698	5537	24378
1988	715.104	79.805	1345	8018	1199.399	42.031	5965	26201
1989	585.651	82.151	1060	6164	758.330	42.981	3893	15912
1990	532.214	82.990	996	5741	888.595	46.370	4105	17729
1991	566.507	88.245	977	5859	878.905	47.862	3724	16731
1992	611.126	91.477	1017	6349	670.331	54.575	2969	12893
1993	548.256	100.152	853	5050	958.501	57.295	3528	15881
1994	499.406	106.578	773	4316	1075.889	61.854	3734	16922
1995	478.919	120.656	668	3662	1093.209	65.002	3417	15488
1996	427.787	131.610	546	3052	1138.427	76.453	3169	14064
1997	657.496	143.157	804	4373	1321.556	74.469	3711	16617
1998	480.660	148.531	591	3038	1449.292	80.190	4098	17087
1999	645.049	156.693	806	3842	1362.256	76.063	4219	16957
2000	960.741	156.464	1248	5679	1211.099	71.104	4139	16322
2001	940.434	143.368	1340	6042	1142.272	56.510	4697	19069
2002	911.717	140.285	1298	5963	966.405	47.775	4658	19064
2003	954.589	137.921	1434	6378	912.870	50.495	4214	17444
2004	935.142	128.719	1524	6712	860.470	54.691	3815	15008
2005	940.138	124.036	1513	7083	835.078	66.553	3042	11963
2006	951.849	125.112	1574	7091	870.872	78.387	2723	10604
2007	1030.043	122.266	1790	7938	880.698	80.388	2802	10760
2008	984.599	111.036	1789	8315	911.409	78.517	2932	11291
2009	835.676	121.224	1463	6480	947.021	82.643	2828	11054
2010	839.659	117.584	1491	6675	988.639	63.701	3732	14618
2011	832.946	118.142	1408	6666	786.855	59.558	3181	12500

The catches in the Tasmanian fishery have been greatly influenced by management changes. Following the introduction of quotas in 1984 the TAC declined to a low of

2100 t by 1989 following which a high proportion of the TAC was taken from the easier to fish east coast. Eventually zonation into east and west coasts was introduced in the year 2000 (Tarbath and Gardner, 2011) and catches jumped from 475 – 600 t in the west back up to 960 t plus. In 2009, a new western zone was defined (Figure 42) with the aim of ensuring the more even distribution of catch across productive areas, and hence the catches dropped in blocks 9 – 12 down to about 830 t (Figure 45).

The western fishery across blocks 9 – 12 has varied over the years from 1985 – 2011 with a range from 427 – 1030 t (Figure 45; Table 15).

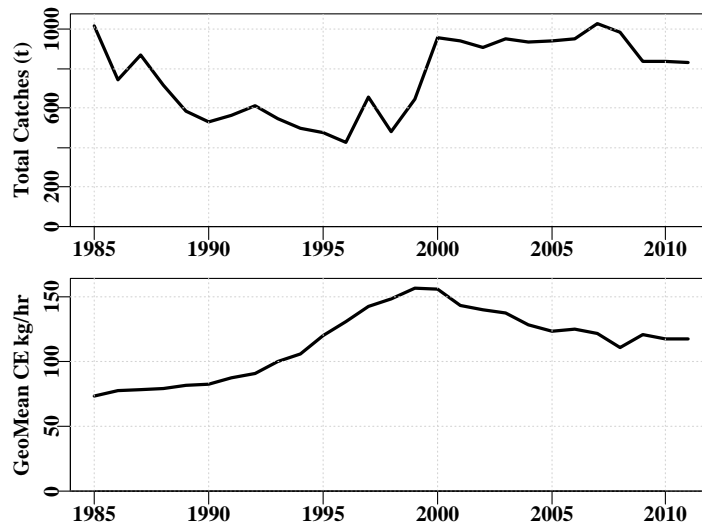


Figure 45. The total catches and geometric mean catch rates from 1985 – 2011 for western blocks 9 – 12 combined.

The fishery across eastern blocks 13 – 31 has varied over the years from 1985 – 2011 with a range from 675.9 – 1515 t (Figure 46; Table 15), although higher catches were reported in the early 1980s.

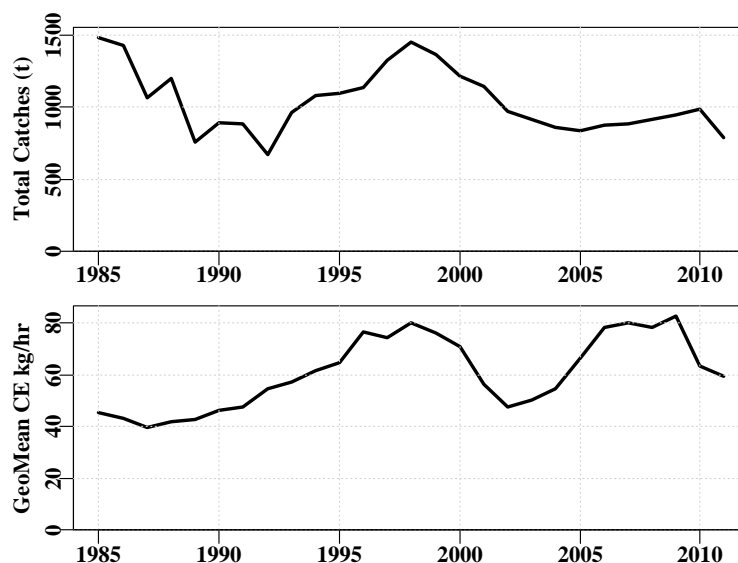


Figure 46. The total catches and geometric mean catch rates from 1985 – 2011 for blocks 13 – 31 combined.

Quotas were introduced in 1984 because the fishery as a whole was in a depleted state with greatly reduced catch rates and widely reported difficulties in fishing from the divers. Following a halving of the catch (or more) and especially on the west coast where catches dropped to record lows, there was a remarkable rebuilding and increase in average catches rates up to an average of ~150 kg/hr. Since the introduction of the zonation and the return of greater catches to the west coast the catch rates have declined steadily although they may be showing signs of stabilizing (**Figure 45**); or they may not as the most recent data from 2012 has begun to be analyzed.

17.6.2 Western Zone: Blocks 9 – 12

17.6.2.1 Catches and Geometric Mean Catch Rates

Table 16. The annual catches in tonnes, and simple geometric mean catch rates in kg.hr for the western zone blocks 9 – 12.

	Catches (t)				Catch Rates Kg/Hr			
	AB09	AB10	AB11	AB12	AB09	AB10	AB11	AB12
1985	229.649	137.437	464.941	155.264	77.5614	75.4524	77.7114	60.1466
1986	129.239	118.139	278.878	186.450	77.5123	81.7796	82.3195	72.2993
1987	248.255	81.373	333.260	192.614	76.1753	91.4311	83.4726	70.7871
1988	155.688	117.373	258.939	158.051	80.2596	90.7288	81.1738	72.2402
1989	119.051	103.657	199.014	131.848	80.8839	99.6795	85.4920	72.2575
1990	94.997	75.844	221.755	123.849	81.9780	87.3252	87.6320	75.4670
1991	100.922	100.263	213.513	137.974	79.6518	98.7046	90.0837	87.2304
1992	90.363	94.637	266.226	158.888	89.3789	96.6243	91.9101	89.7735
1993	109.438	64.644	197.059	176.932	101.8042	91.9178	102.5804	99.8275
1994	76.747	60.038	201.253	160.192	114.0280	116.2215	105.9805	101.7612
1995	43.565	68.021	185.453	181.753	125.5182	149.0145	116.7629	116.8182
1996	59.341	74.687	145.252	148.486	149.2423	134.6163	136.5236	122.1172
1997	139.816	66.447	224.491	226.742	153.2317	160.7452	140.8305	136.9524
1998	78.158	46.635	163.401	192.466	173.7265	167.6026	157.6662	132.9077
1999	115.200	58.437	220.336	251.076	197.5366	176.2590	164.7127	138.1046
2000	205.208	147.541	326.447	281.545	167.1500	197.3020	163.4860	134.5157
2001	186.242	151.981	312.264	289.947	148.5943	176.9556	149.4015	126.6937
2002	173.517	142.514	359.747	235.939	151.3819	176.7601	153.9779	112.2550
2003	142.442	238.485	344.452	229.210	142.1012	163.4455	141.1435	117.4568
2004	130.284	181.442	373.852	249.564	138.5316	159.3362	142.9872	103.0186
2005	91.799	148.834	388.924	310.581	147.2086	146.4327	137.1594	103.9899
2006	141.943	197.764	383.845	228.297	139.6570	149.3214	128.8361	103.3318
2007	177.717	231.195	353.816	267.315	144.8986	146.5942	118.4374	103.6969
2008	156.175	178.246	345.043	305.135	144.9297	123.1422	111.0885	97.2840
2009	154.578	110.262	243.717	327.119	168.7249	122.9798	122.1088	107.6895
2010	157.747	157.809	243.556	276.643	152.9262	148.3181	119.4077	96.2778
2011	171.130	158.879	246.724	256.213	161.0472	138.0732	108.4108	102.0170

When the four blocks in the south west are considered separately the yields from each area differ substantially through time with block 11 generally producing more catch than block 12 and well above blocks 9 and 10. Now that catches are more constrained by the recently reduced TAC, blocks 11 and 12 are more equal as are blocks 9 and 10, although there remains a difference of about 80-90 t between the two pairs (**Table 16; Figure 47**). Despite this disparity in catches the trends in the catch rates from the four blocks remain surprisingly similar.

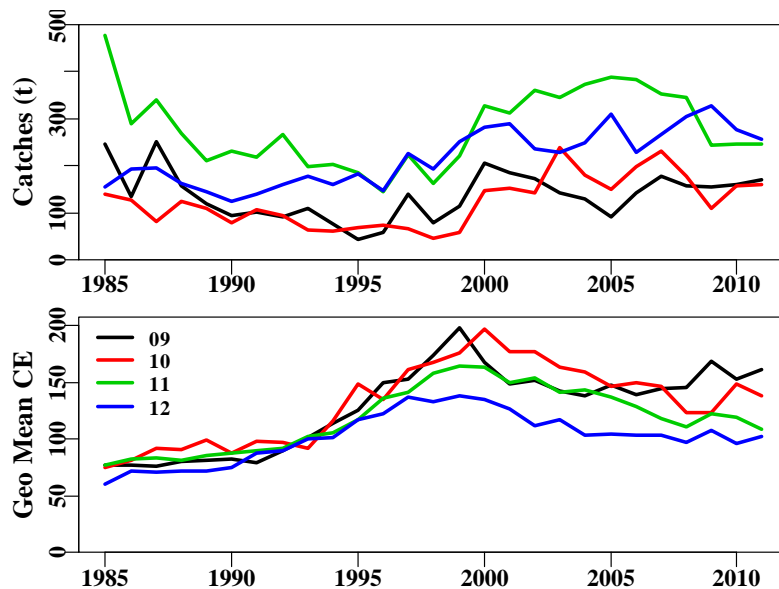


Figure 47. The time series of catches and of geometric mean catch rates for each of the four blocks (blocks 9 – 12) in Tasmania’s south-western region.

17.6.2.2 Statistical Standardizations

The standardization of the whole zone demonstrates that the standardization only has a relatively minor effect on the perceived trends in catch rates (**Figure 48; Table 17; Table 18**). Despite this minor effect the standardization accounts for almost 40.5% of available variation, most of which is related to between year changes (23.5%) and to the divers doing the fishing (12.9%), the month factor added a small amount (3%) and block, with its interaction with month were very minor (**Table 17**).

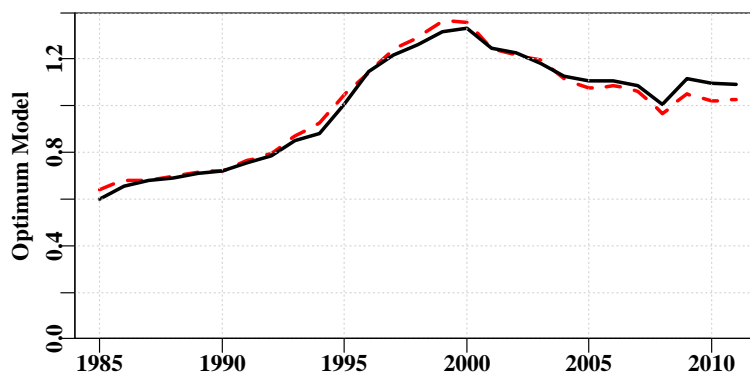


Figure 48. A comparison of the simple geometric mean catch rates (scaled so each time series has a mean of 1.0) with the optimum model from a statistical standardization for all four blocks together. The optimum model is the solid black line, the geometric mean is the dashed red line.

Table 17. The model selection results for the analysis relating to the complete south western coast. The various factors are cumulative in the statistical models. AIC is the Akaike's Information Criterion, RSS is the residual mean square, MSS is the model mean square, Nobs is the number of observations, Npars is the number of parameters in the statistical model, adj_r2 is the adjusted r^2 , essentially the variation accounted for with allowance for the number of parameters, and %change is change in the adj_r2.

	Year	Diver	Block	Month	Month:Block
AIC	-57101	-62971	-64583	-65055	-65091
RSS	5978	4921	4688	4619	4605
MSS	1849	2906	3139	3208	3222
Nobs	33287	33287	33287	33287	33287
Npars	27	330	333	344	377
adj_r2	23.563	36.498	39.505	40.376	40.498
%Change		12.936	3.006	0.872	0.122

Despite the standardization only changing the trend slightly it does so differently either side of the introduction of zones in 2000. Prior to 2000 the standardized catch rates are sometimes slightly lower than the geometric means while after they are somewhat higher, getting more so in more recent years with the last three years being about 7% higher than the unstandardized catch rates (**Figure 48**). The similarity of the catch rate trends in the four blocks becomes visually more apparent when each time series is scaled to a mean of 1.0 (**Figure 49**).

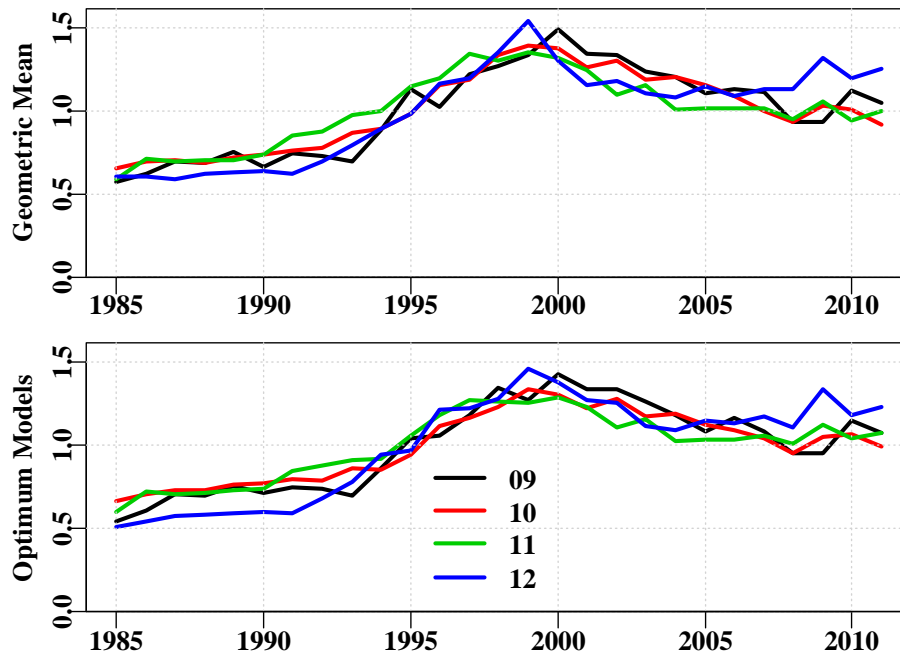


Figure 49. A comparison of the simple geometric mean catch rates (scaled so each time series has a mean of 1.0) with the optimum models produced for each block from a statistical standardization.

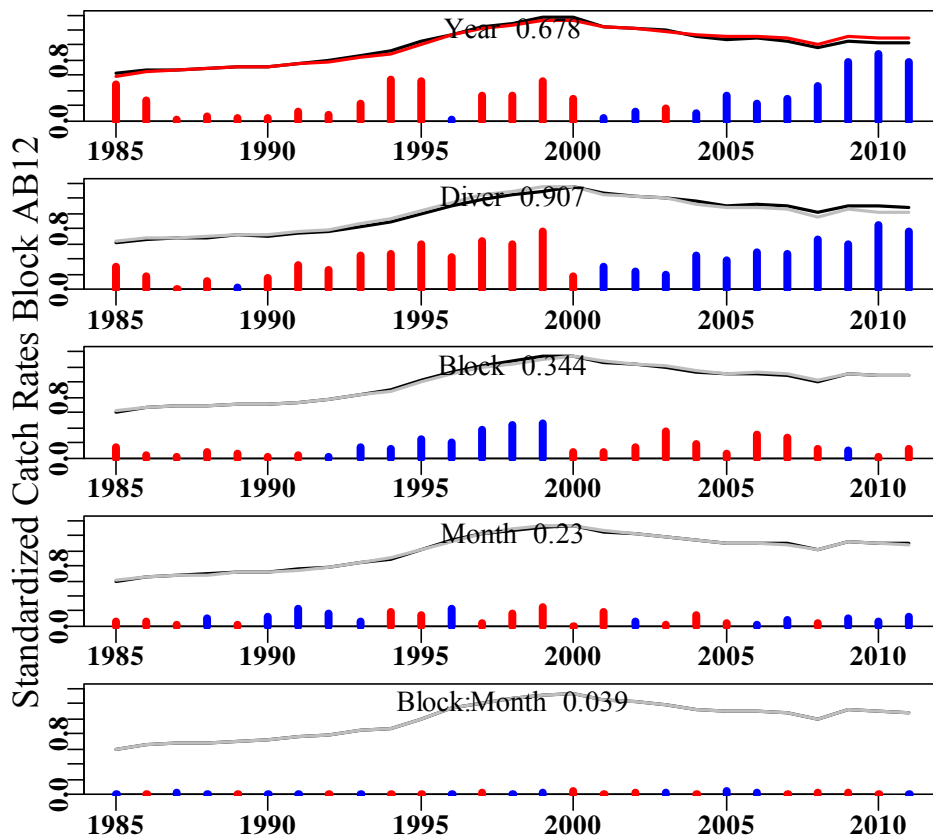


Figure 50. The effect of each factor on the western catch rate trend cumulated across all factors. The number in each graph is the sum of squared differences as an index of change. The grey line in each case represents the cumulative effect of all factors above each case, while the black line illustrates the change by the factor concerned.

When the optimum standardized models for each of the four blocks are compared with the analysis for all four blocks together (**Figure 48, Figure 49**) the variation about the average becomes apparent. The effect of the standardizations is mainly to make the trends less variable. It is important to note that even though the catches in the different areas are variable and different between blocks, the catch rates, either standardized or otherwise, for all four blocks, follow essentially the same trends through time (**Figure 47, Figure 49**).

Each of the factors is statistically significant (**Table 17**), although for the last two factors this may be more a reflection of the large number of observation rather than any major influence over catch rates once the earlier factors have had their effect. The different factors each has an effect on the catch rate trend but the first three, Year, Diver, and Block account for most of the variation in the catch rate trend. The Diver factor decreases catch rates prior to 2000 and increase it afterwards, while the Block factor generally does the opposite (**Figure 50**). The month and block:month interaction terms appear to represent mainly noise in the trend.

Table 18. Scaled standardization coefficients for each model relating to all data combined plus the optimum model year coefficients for the four Block based models.

Year	Model components for all data combined					Optimum models for each block's data			
	GeoTot	Diver	Month	Block	MthBlk	Mth9	Mth10	Mth11	Mth12
1985	0.6389	0.6145	0.6089	0.5975	0.5984	0.5088	0.5431	0.6688	0.5996
1986	0.6795	0.6656	0.6609	0.6578	0.6571	0.5468	0.6079	0.7061	0.7241
1987	0.6802	0.6792	0.6774	0.6763	0.6786	0.5749	0.7083	0.7307	0.7069
1988	0.6932	0.6841	0.6897	0.6867	0.6875	0.5859	0.6961	0.7327	0.7126
1989	0.7136	0.7162	0.7141	0.7110	0.7105	0.5954	0.7538	0.7643	0.7327
1990	0.7209	0.7082	0.7163	0.7178	0.7181	0.6000	0.7163	0.7738	0.7386
1991	0.7665	0.7390	0.7568	0.7558	0.7564	0.5962	0.7522	0.7968	0.8491
1992	0.7946	0.7727	0.7847	0.7880	0.7872	0.6795	0.7383	0.7885	0.8776
1993	0.8700	0.8333	0.8365	0.8495	0.8508	0.7835	0.7031	0.8660	0.9127
1994	0.9258	0.8860	0.8738	0.8813	0.8802	0.9412	0.8666	0.8524	0.9220
1995	1.0481	0.9977	0.9926	1.0052	1.0050	0.9669	1.0467	0.9486	1.0560
1996	1.1433	1.1076	1.1242	1.1438	1.1451	1.2155	1.0580	1.1158	1.1802
1997	1.2436	1.1904	1.1922	1.2178	1.2164	1.2257	1.1816	1.1695	1.2721
1998	1.2903	1.2406	1.2304	1.2624	1.2625	1.2790	1.3437	1.2333	1.2645
1999	1.3611	1.2979	1.2821	1.3150	1.3180	1.4568	1.2735	1.3403	1.2570
2000	1.3591	1.3449	1.3447	1.3378	1.3342	1.3795	1.4253	1.3088	1.2900
2001	1.2453	1.2706	1.2560	1.2500	1.2490	1.2694	1.3417	1.2231	1.2287
2002	1.2186	1.2389	1.2426	1.2323	1.2292	1.2586	1.3372	1.2833	1.1085
2003	1.1980	1.2135	1.2095	1.1817	1.1839	1.1163	1.2656	1.1762	1.1569
2004	1.1181	1.1554	1.1441	1.1278	1.1270	1.0937	1.1810	1.1943	1.0235
2005	1.0774	1.1098	1.1071	1.1016	1.1051	1.1487	1.0852	1.1255	1.0386
2006	1.0867	1.1277	1.1282	1.1039	1.1060	1.1372	1.1622	1.0904	1.0389
2007	1.0620	1.1017	1.1078	1.0856	1.0855	1.1766	1.0832	1.0414	1.0567
2008	0.9645	1.0195	1.0148	1.0062	1.0036	1.1125	0.9538	0.9530	1.0136
2009	1.0530	1.1032	1.1105	1.1202	1.1179	1.3383	0.9524	1.0507	1.1227
2010	1.0214	1.0926	1.0942	1.0965	1.0954	1.1830	1.1511	1.0677	1.0422
2011	1.0262	1.0892	1.0998	1.0903	1.0915	1.2300	1.0720	0.9982	1.0739

17.6.3 Eastern Zone Blocks 13 – 31

17.6.3.1 Catches and Geometric Mean Catch Rates

The eastern fishery was being exploited even more strongly than on the west and the high catches in 1985 followed on from even larger reported catches in the early 1980s, and there are reports of further unreported catches happening prior to the start of quotas. The obvious depletion across the State led to the introduction of quotas in 1985 starting at a TAC of 3,806 t which declined to 2,076 t in 1989 and stabilized at 2,100 t in 1990 (Tarbath and Gardner, 2011). Reducing the eastern catches to the extent which this entailed allowed the resource to recover, as evidenced by the gradual increase in catch rates from about 1987 – 1998. These increases were matched from 1993 by continually increasing catches taken from the eastern coast (**Figure 46, Figure 51**).

The catches taken from the four areas up the east coast differ in that those from blocks 13 – 14 are often twice that from the other areas, with the general pattern being that

catches decrease the further north up the east coast the fishing occurs. The simple catch rates, however, are all remarkably similar in their trends through time, with the whole east coast exhibiting the strong oscillations apparent since 1994 (**Figure 51**). The frequency distribution of catch rates in any single year all appear to have similar distributions even though the mean values can differ somewhat (**Figure 52**).

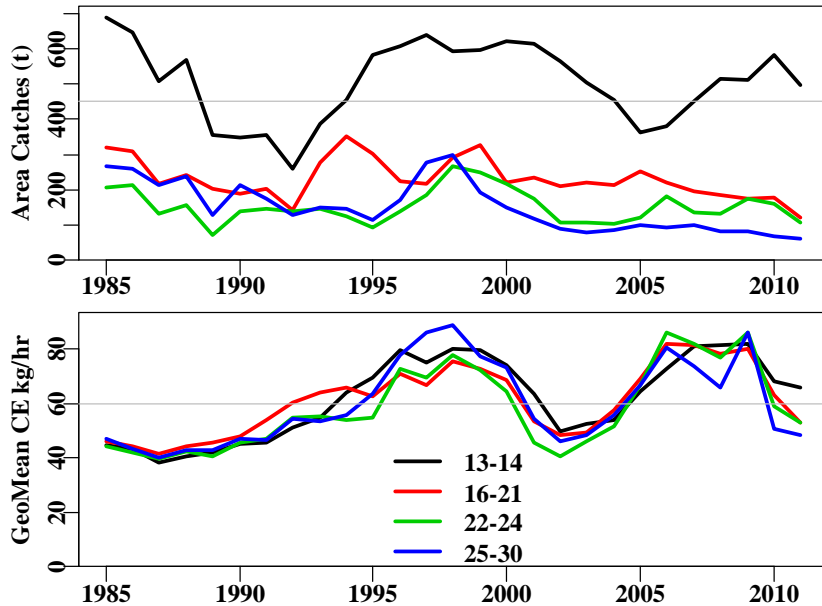


Figure 51. The unscaled catches and geometric mean catch rates for the four areas defined by the blocks identified in the legend.

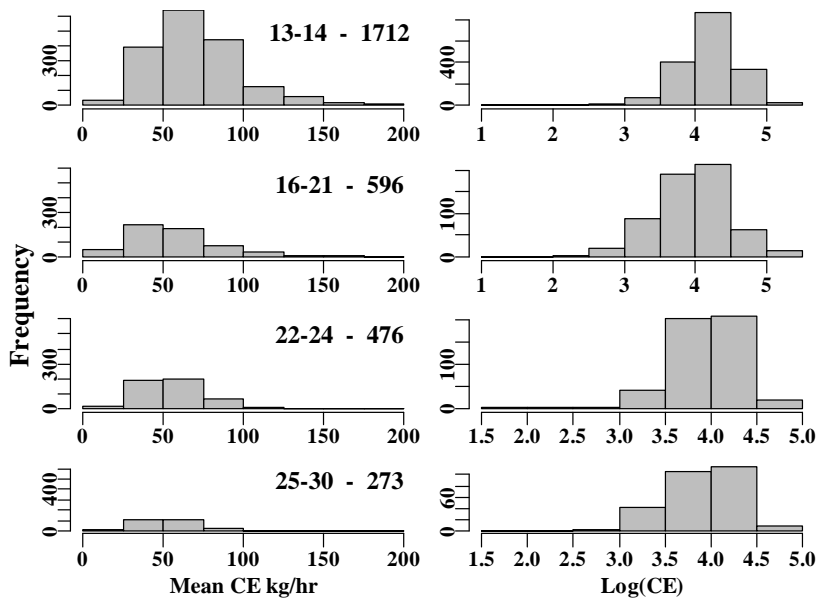


Figure 52. The frequency distributions of catch rates in 2011 (both raw catch rates and log-transformed catch rates) observed on the east coast. The legend in each case describes the blocks the row refers to and the number of records reported in 2011.

There have been numerous management changes on the east coast including changes to the legal minimum length and the TAC (Tarbath & Gardner, 2011). While the catches

from the most productive area exhibits similar oscillations to the catch rates the more northerly areas have experienced lower catches since the recent reductions in TAC on the east coast (**Figure 51**).

17.6.3.2 Statistical Standardization

The standardization of the whole eastern coast demonstrates that, like the west coast, the standardization only has a relatively minor effect on the perceived trends in catch rates (**Figure 48**, **Figure 53**; **Table 18**). As with the west coast the standardization lowers the catch rates prior to zonation and increases it after zonation.

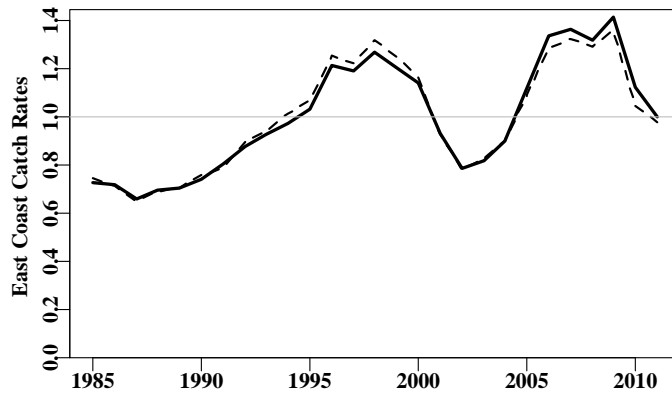


Figure 53. A comparison of the simple geometric mean catch rates (scaled so each time series has a mean of 1.0) with the optimum model from a statistical standardization for all eastern coast blocks together. The optimum model is the solid black line, the geometric mean is the dashed black line

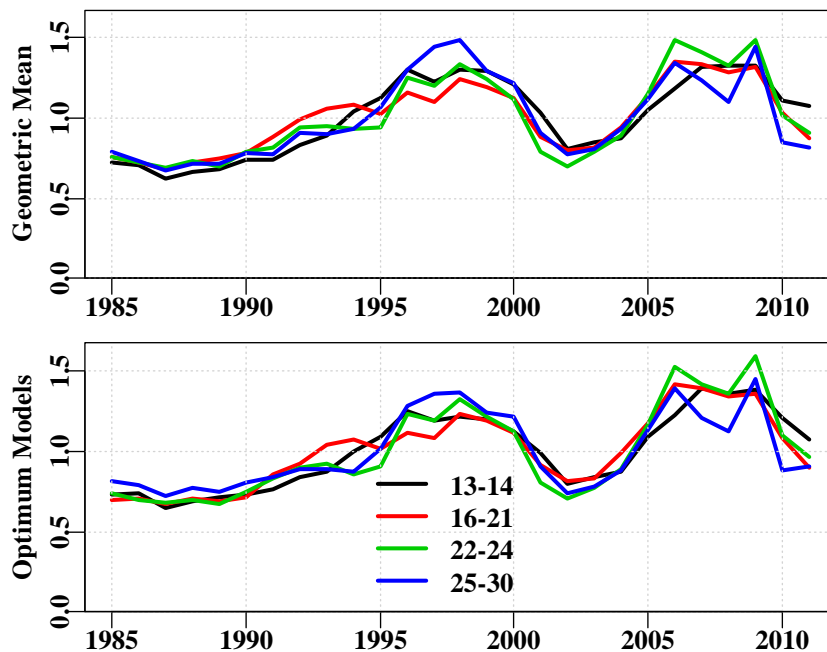


Figure 54. A comparison of the simple geometric mean catch rates (scaled so each time series has a mean of 1.0) with the optimum models produced for each area from a statistical standardization of each collection of blocks separately. Similarly, when the catch rates for the individual areas are compared both the geometric

mean and optimum statistical models remain very similar indicating that catch rates are reflected consistently right up the eastern coastal fishery (**Figure 54; Table 19**).

Table 19. Scaled standardization coefficients for each east coast model relating to all data combined plus the optimum model year coefficients for the four Area/Region based models.

Year	GeoMean	Diver	Month	Block	Blk:Mth	Opt_R1	Opt-R2	Opt-R3	Opt-R4
1985	0.7453	0.7174	0.7065	0.7291	0.7300	0.7318	0.6951	0.7385	0.8122
1986	0.7126	0.6939	0.6940	0.7182	0.7205	0.7374	0.7016	0.6975	0.7866
1987	0.6520	0.6429	0.6423	0.6601	0.6617	0.6496	0.6760	0.6780	0.7188
1988	0.6903	0.6791	0.6756	0.6958	0.6980	0.6918	0.7022	0.6939	0.7759
1989	0.7059	0.6972	0.6865	0.7037	0.7039	0.7112	0.6918	0.6746	0.7475
1990	0.7616	0.7340	0.7238	0.7416	0.7409	0.7295	0.7157	0.7454	0.8088
1991	0.7861	0.7856	0.7871	0.8075	0.8069	0.7597	0.8562	0.8292	0.8375
1992	0.8964	0.8709	0.8639	0.8797	0.8792	0.8370	0.9255	0.9002	0.8895
1993	0.9410	0.9222	0.9164	0.9273	0.9257	0.8681	1.0386	0.9184	0.8904
1994	1.0159	0.9798	0.9677	0.9754	0.9718	0.9970	1.0688	0.8545	0.8731
1995	1.0676	1.0365	1.0252	1.0325	1.0328	1.0907	1.0143	0.9068	1.0107
1996	1.2557	1.2264	1.2208	1.2170	1.2131	1.2520	1.1166	1.2301	1.2806
1997	1.2231	1.2080	1.2075	1.1912	1.1926	1.1910	1.0784	1.1865	1.3526
1998	1.3170	1.2883	1.2775	1.2641	1.2658	1.2167	1.2280	1.3230	1.3641
1999	1.2493	1.2207	1.2211	1.2087	1.2045	1.2017	1.1883	1.2157	1.2417
2000	1.1678	1.1415	1.1524	1.1398	1.1388	1.1263	1.1184	1.1251	1.2148
2001	0.9281	0.9189	0.9277	0.9308	0.9344	0.9878	0.9121	0.8069	0.9051
2002	0.7847	0.8027	0.8003	0.7892	0.7875	0.7958	0.8169	0.7050	0.7403
2003	0.8293	0.8336	0.8325	0.8215	0.8205	0.8424	0.8266	0.7709	0.7777
2004	0.8982	0.9081	0.9139	0.9027	0.9027	0.8761	0.9862	0.8857	0.8816
2005	1.0931	1.1233	1.1230	1.1186	1.1174	1.0860	1.1710	1.1642	1.1307
2006	1.2874	1.3183	1.3366	1.3354	1.3368	1.2227	1.4158	1.5236	1.3920
2007	1.3203	1.3732	1.3818	1.3674	1.3630	1.3890	1.3865	1.4151	1.2086
2008	1.2896	1.3315	1.3352	1.3106	1.3171	1.3548	1.3368	1.3527	1.1186
2009	1.3573	1.4036	1.4314	1.4144	1.4119	1.3798	1.3549	1.5899	1.4486
2010	1.0462	1.1066	1.1281	1.1227	1.1228	1.2025	1.0781	1.1006	0.8821
2011	0.9782	1.0359	1.0210	0.9950	1.0000	1.0715	0.8997	0.9679	0.9097

Because of the strong oscillations across the years each standardization was dominated by the Year factor followed by the Diver factor, the block was very slightly more influential on the trend than Month although the factors Month, Block, and their interaction were all relatively minor in their influences (**Table 20; Figure 55**). When the Month factor is estimated prior to the Block factor in the models its influence declines to 1.6% and Block increases to 1.8%. The influence of Block was positive up to 1995 and negative after then, but its effects although statistically significant remained small on the general trend. Given the large number of observations even the interaction term is deemed significant even though it increased the number of parameters by 165 and only increased the r^2 by 0.45%.

Table 20. The model selection results for the analysis relating to the complete Eastern coast. The various factors are cumulative in the statistical models. AIC is the Akaike's Information Criterion, RSS is the residual mean square, MSS is the model mean square, Nobs is the number of observations, Npars is the number of parameters in the statistical model, adj_r2 is the adjusted r^2 , essentially the variation accounted for with allowance for the number of parameters, and %change is change in the adj_r2.

	Year	Diver	Block	Month	Block:Month
AIC	-169510	-190508	-193402	-196410	-197040
RSS	21276	17334	16862	16385	16237
MSS	5925	9867	10340	10816	10964
Nobs	105748	105748	105748	105748	105748
Npars	27	362	377	388	553
adj_r2	21.764	36.057	37.792	39.543	39.995
%Change	0.000	14.292	1.735	1.751	0.452

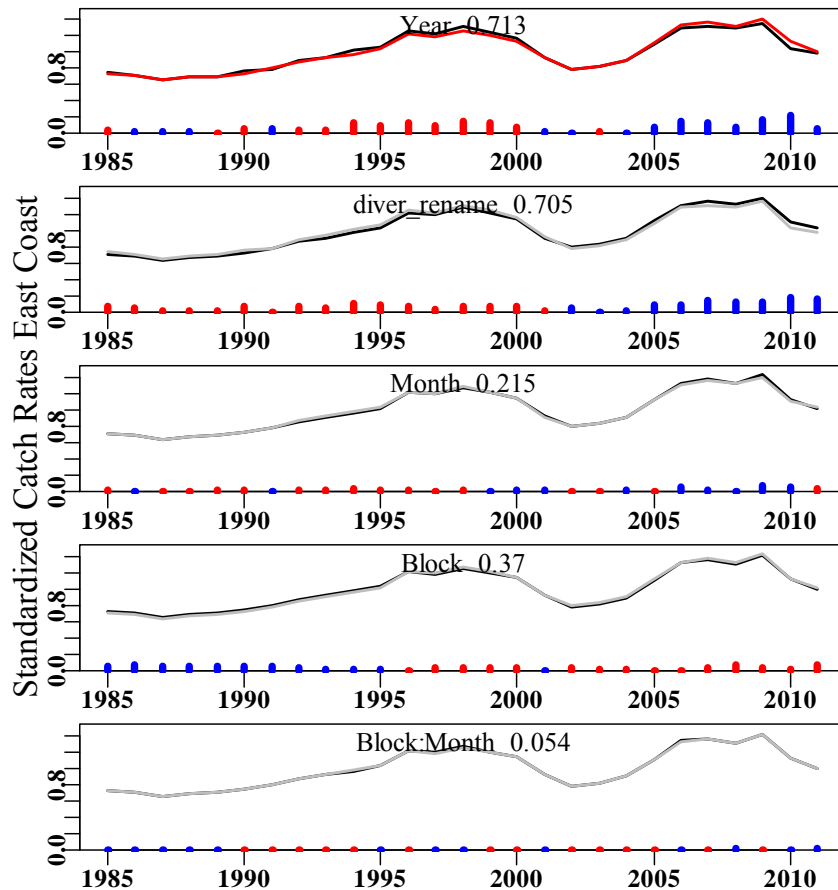


Figure 55. Impact plot for the analysis relating to the whole east coast. Each plot illustrates the effect on the trend exhibited by the scaled geometric mean catch rates of the identified factor. The red line in the top plot is the influence of Year while the black line is the geometric mean. In all other plots the grey line is the result of the previous model above and the black line is the influence of the given factor. The number in each case is the squared difference between the two lines summed across years. If the bars are red the effect is negative and if blue it is positive.

17.6.4 Do Catch Rates Reflect Stock Status?

The relationship between catches and subsequent catch rates as determined using time lagged time series from the western zone indicated significant relationships with time lags from 4 – 9 years with the optimum lag being 7 years, although 6 and 8 years are also highly significant (Figure 56; Table 21). This differs from the eastern zone with an optimum lag of 5 years although a 4 year lag is very similar (Figure 57; Table 22).

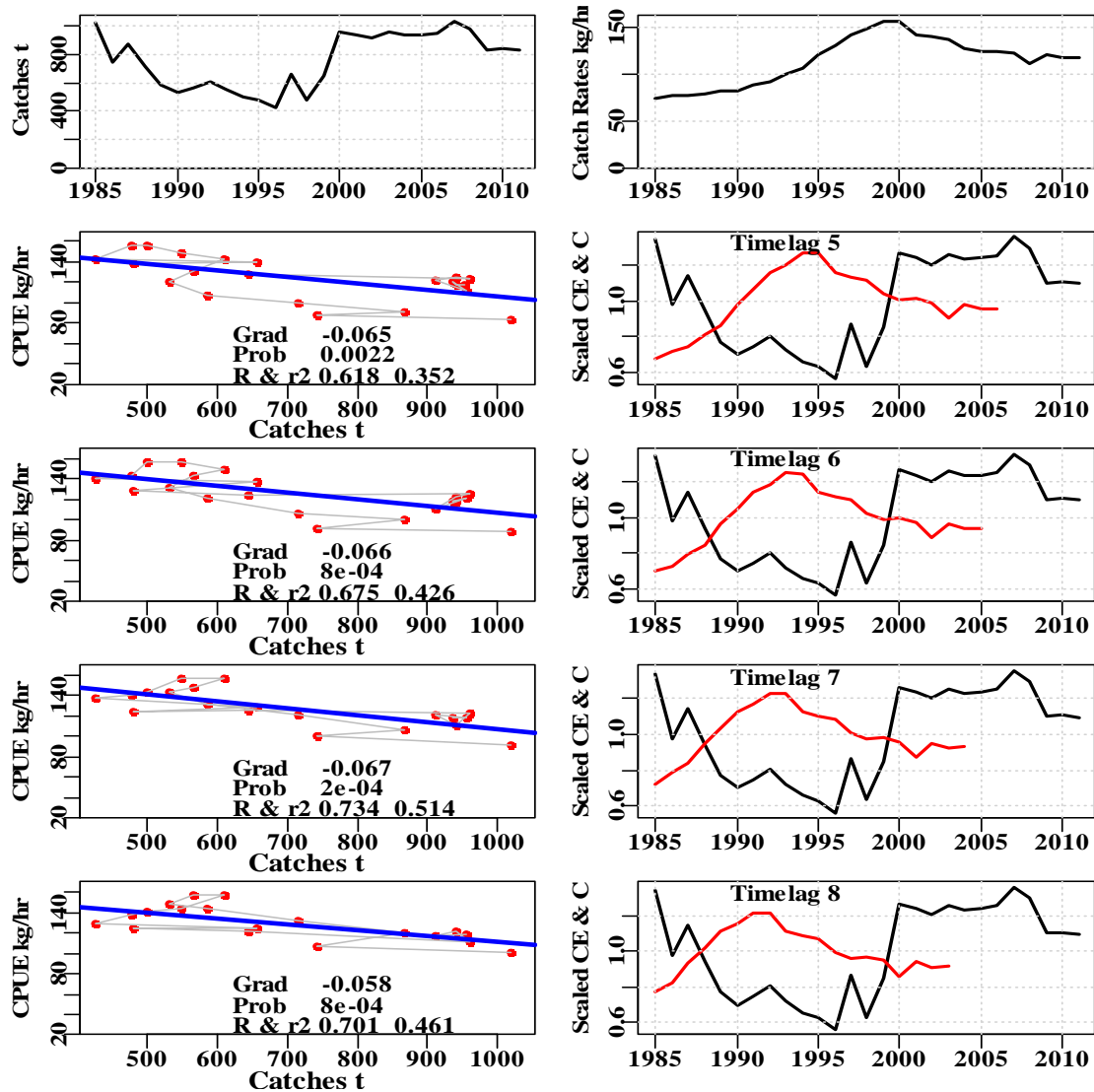


Figure 56. The relationship between catches (black lines) and catch rates (red lines) for the western blocks 9 – 12 when the catch rates are lagged backwards by different numbers of years. The numbers on the lower four graphs are the gradient of the relationship, the probability that it is statistically significant, and the correlation coefficient and the adjusted r^2 . A time-lag of seven means that the catch rates in 2002 would be matched with the catches in 1995.

Table 21. The regression modelling of the correlation between time lagged catches and subsequent catch rates for the western coast blocks 9 - 12 from 1985 – 2011. The Gradient is the gradient of each regression and the adj-r² is the adjusted r². The highlighted lines are significant with the optimal model being with a time-lag of 7.

Time-Lag	Gradient	Correlation	adj r ²	Probability	Observations
0	0.0201	0.1521	-0.0159	0.44886	27
1	-0.0042	0.0335	-0.0405	0.87100	26
2	-0.0223	0.1869	-0.0070	0.37101	25
3	-0.0376	0.3337	0.0710	0.11102	24
4	-0.0534	0.4943	0.2084	0.01650	23
5	-0.0648	0.6184	0.3515	0.00216	22
6	-0.0658	0.6745	0.4263	0.00080	21
7	-0.0666	0.7345	0.5139	0.00023	20
8	-0.0583	0.7010	0.4614	0.00083	19
9	-0.0409	0.5161	0.2205	0.02835	18
10	-0.0160	0.2106	-0.0193	0.41712	17

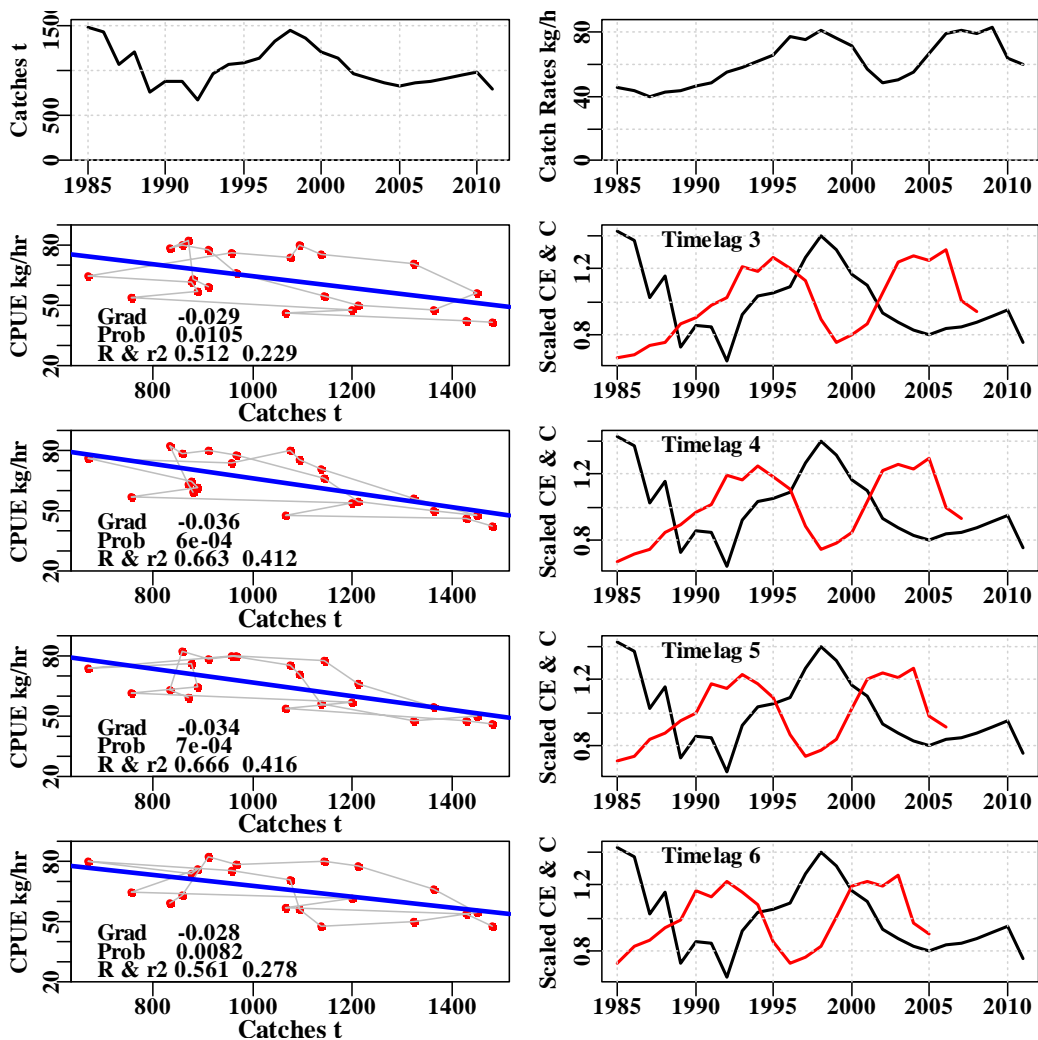


Figure 57. The relationship between catches (black lines) and catch rates (red lines) for the eastern blocks 13 – 31 when the catch rates are lagged backwards by different numbers of years. The numbers on the lower four graphs are the gradient of the relationship, the probability that it is statistically significant, and the correlation coefficient and the adjusted r². A time-lag of five means that the catch rates in 2000 would be matched with the catches in 1995.

Table 22. The regression modelling of the correlation between time lagged catches and subsequent catch rates for the eastern coast blocks 13 – 31 from 1985 – 2011. The Gradient is the gradient of each regression and the adj-r² is the adjusted r². The highlighted lines are significant with the group from time-lags 3 – 6 having negative gradients but 4 and 5. The significant relationship at a lag of 10 years is a reflection of the most significant relationship at 5 years.

Time-Lag	Gradient	Correlation	adj_r ²	Probability	Observations
0	0.0051	0.0804	-0.0333	0.69012	27
1	-0.0066	0.1054	-0.0301	0.60822	26
2	-0.0185	0.3038	0.0528	0.13980	25
3	-0.0294	0.5121	0.2287	0.01052	24
4	-0.0361	0.6627	0.4125	0.00057	23
5	-0.0344	0.6662	0.4160	0.00071	22
6	-0.0277	0.5608	0.2784	0.00818	21
7	-0.0173	0.3640	0.0843	0.11465	20
8	-0.0008	0.0171	-0.0585	0.94474	19
9	0.0113	0.2402	-0.0012	0.33700	18
10	0.0248	0.5266	0.2291	0.02987	17

The range of catches in the western blocks was from a minimum of 427 t to a maximum of 1030 t, while in the eastern blocks the range was from 670 t to 1482 t. The catch rates doubled between the minimum and maximum in the east (39.6 – 82.6 kg/hr; a ratio of 1:2.08) while catch rates slightly more than doubled from the minimum to the maximum (73.5 – 156.7 kg/hr; a ratio of 1:2.13).

The trends in catch rates in the eastern blocks indicates the stock started off low, increased in size, decreased again, recovered again, only to begin to decline again. In the western blocks the stock also started low but recovered up to a high in 2000 and has since declined.

The relationship between catches and subsequent catch rates is stronger in the western blocks than in the eastern blocks, with the correlations being higher and accounting for more of the available variation. This can be perceived visually by the tighter scatter of data about the regression lines for the western blocks.

When these trends are combined with the relationships between catches and time lagged catch rates, this constitutes evidence that time series of catch rates in this abalone fishery can be informative about the stock dynamics and its status.

17.7 Discussion

Harvesting abalone involves hand-gathering (although the use of long poles with terminal hooks has been reported for the early fishery in British Columbia (Sloan and Breen, 1988), and the use of spears to access abalone deeper than could be reached from the surface was reported from the nineteenth century fishery in Tasmania (Harrison, Pers. Comm.). The ability of divers to alter their behaviour and maintain catch rates as local

stock size declines is the underlying reason that catch rates from commercial log books are generally considered unreliable indices of relative stock abundance. For example, during an extensive stock decline that occurred across the Tasmanian east coast during the early 2000s, one strongly performing diver didn't agree that a decline was occurring because, as he reported, he found he could maintain his catch rates by simply swimming faster and further. It might be expected therefore that the divers conducting the fishing would be very influential on catch rates, and variation in the divers fishing might also act to confuse any signal about relative abundance through time.

The current Tasmanian abalone fishery started in the 1960s and developed through the 1970s into a reported annual fishery of over 4,500 t (Tarbath and Gardner, 2012). The effect of the changes in catches and effort through time was to impose contrasting effort and related catch levels across the fishery at different times which, unintentionally led to the generation of information concerning how the stocks responded to an array of widely different fishing mortality levels (this is known as the fishery data exhibiting contrast). Thus, the catch levels reached in the late 1970s and early 1980s led to reductions in the stock which were obvious to the divers. This serious reduction led to the introduction of a quota system in 1985 with an initial Total Allowable Catch (TAC) of 3,806 t, however, by only 1989 this had been reduced by 45% until the TAC was only 2,076 t, which eventually stabilized at 2,100 t from 1990 to 1997. Such large changes to fishing effort and its related mortality provide the best opportunity for any fishery statistics available (catch rates and distribution of effort) to exhibit any responses that they are capable of undergoing.

A number of unexpected outcomes derived from the consideration of catch rates within two major sections of the Tasmanian abalone fishery. Even though catches from different areas within a zone were often markedly different the catch rates exhibited across those areas, even across the whole of the east coast, followed very similar trends. This suggested that the relative abundance across zones varies in the same way. The absolute abundance certainly varies but how the different absolute abundances increase and decrease appears to follow similar patterns throughout each zone. This suggests that recruitment in abalone stocks is more predictable than originally assumed. It may be highly variable ranging from high to low, but generally it must be following similar trends of highs and lows across relatively large areas.

Fortunately for the stock assessments, the Tasmania abalone stocks have undergone some large changes in recent decades such that catches have varied whereby the maximum is just over double the minimum. Catch rates have exhibited similar ranges. This means the stocks have experienced rather different levels of fishing mortality and different levels of stock abundance. Such large changes are ideal for characterizing the potential responses of the stock to such variation. This is known as the fisheries data exhibiting contrast. This is fortunate for stock assessments because they constrain the possible dynamics much more than situations where catches and catch rates have, say, only declined or increased.

Finally, the examination of the data to determine if there were any time-lagged relationship between catch levels and later catch rates found significant correlations with peak correlation sat 7 years on the west coast and 5 years on the east (although a small range

of the surrounding years were also significant). This shows that the stocks have been responding to fishing in the manner expected if catch rates are reflective of the relative abundance of the exploitable biomass.

Each of these outcomes indicates that the use of abalone fishery dependent catch rates can certainly be informative about the relative status of the stock and that catch rates could indeed be used as an empirical performance measure. The fact that there are time-lags, however, means that there could be real delays in management actions (controlling the catch levels) influencing the performance measure being used to recommend those management actions. This is not surprising as changing the catches are likely to be influential on the amount of spawning biomass available, which will influence the recruitment levels. But for that to influence catch rates the new juvenile abalone need to grow through the LML and become available to the fishery. It is also not surprising that there were a range of years over which a significant influence of catches on catch rates could be detected. The growth of abalone is not deterministic and some animals would take less and other more than an average number of years to growth up to and through the LML.

17.8 Catch Length Structure

17.8.1 Changes in Commercial Length Frequencies

17.8.1.1 West Coast

When the commercial length frequency of catches in subblocks 9A – 13B (the new western zone, as defined from 2009 – 2012) are compared through time, only relatively minor differences are exhibited through the years 2008 – 2011, with rather a larger change in 2012 (**Figure 58**, **Figure 59**).

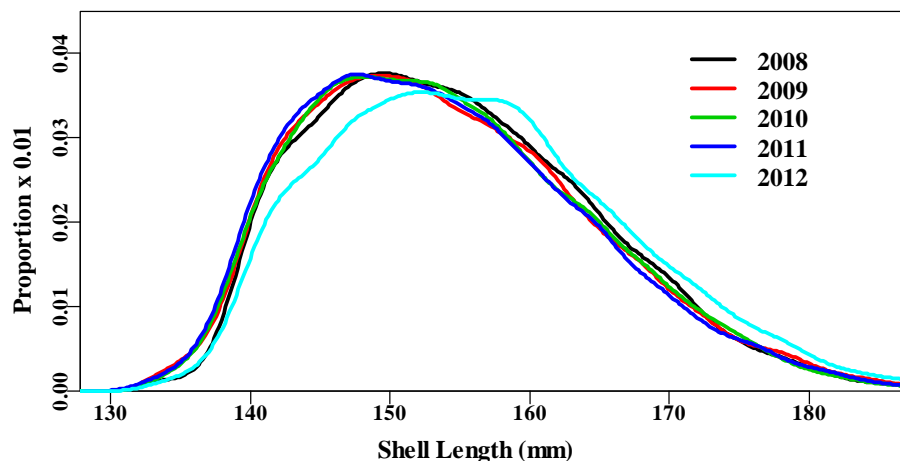


Figure 58. The distributions of the density estimates for subblocks 9A to 13B for the five years 2008 – 2012.

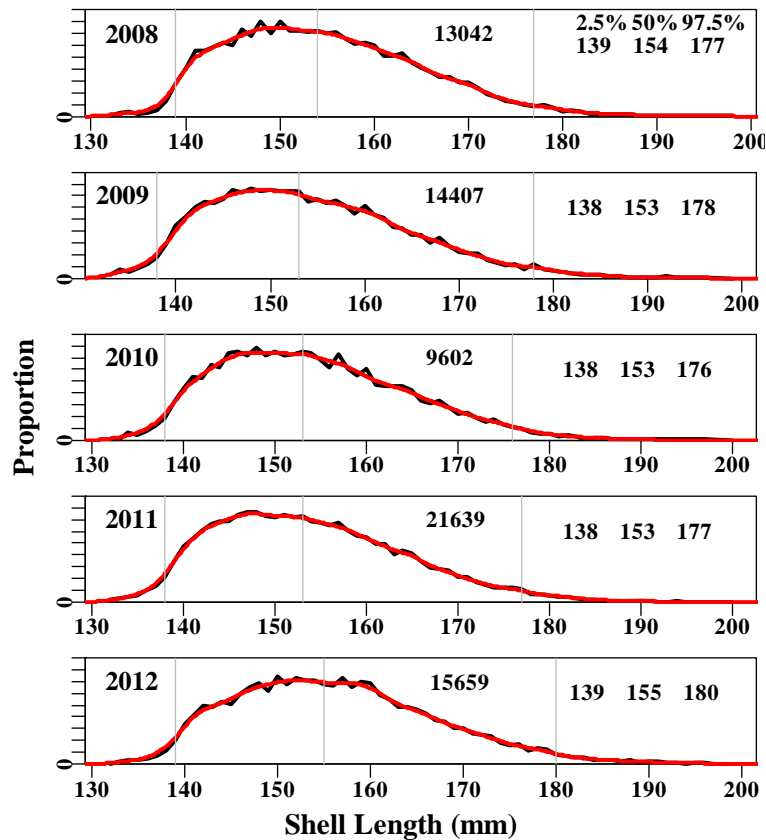


Figure 59. The length frequency of commercial catches from subblocks 9A – 13B each year from 2008 – 2012. The black lines are the observed frequencies while the red lines are the empirical density profile. The y-axis maximum in each case is 0.045. The number above 170mm is the number of observations each year; the percentages are each year's quantiles.

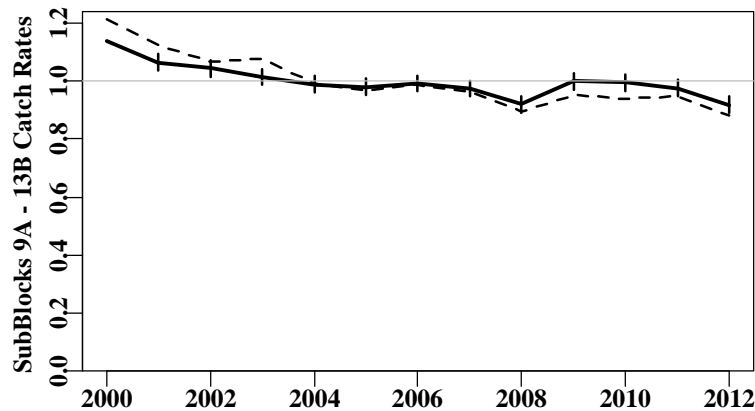


Figure 60. Standardized catch rates for subblocks 9A – 13B combined, over the years 2000 – 2012. The dotted line is the geometric mean while the solid line represents the optimum standardized model.

The inner 95% of each distribution remains basically the same each year with a very slight decline during 2009 – 2011, ranging between 138 – 178, but moving slightly upwards in 2012 to range between 139 - 180. However, the quantiles are somewhat misleading with respect to the changes that occurred in 2012. The shift to the right (to larger sizes) appears relatively marked when compared to the distributions of the previous

four years (**Figure 58**). The catch rates in 2008 and in 2012 were somewhat lower than the other years, and both those years have size distributions that are larger than the other years, especially in 2012, but otherwise there appears to be little trend in the catch rates during those five years (**Figure 60**).

17.8.1.2 Eastern Zone: Actaeons

When we examine only the data from subblocks 13C, 13D, and 13E, which comprises the Actaeon Island area, then similar to that observed in the west, only relatively minor differences are exhibited through the years 2009 – 2011, with rather a larger changes relative to 2008 and 2012 (**Figure 61**, **Figure 62**, and **Figure 63**).

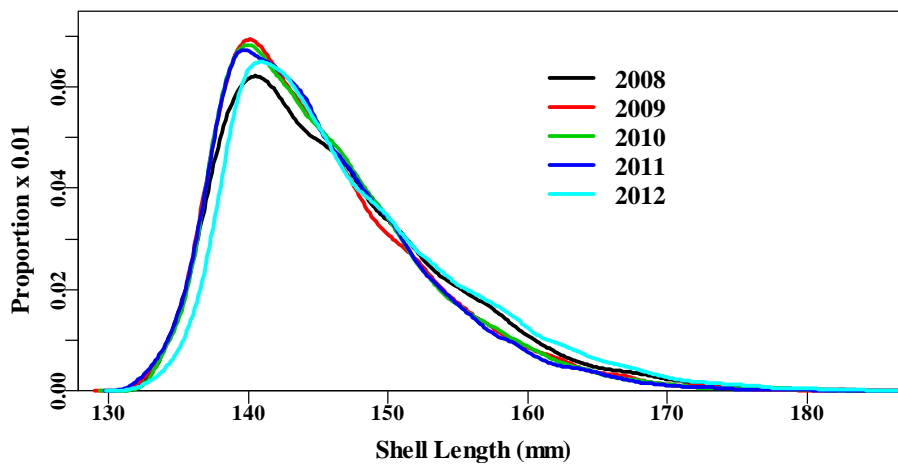


Figure 61. The distributions of the density estimates for subblocks 13C, 13D, and 13E for the five years 2008 – 2012. The differences appear relatively minor but these are sufficient to indicate the changes from year to year brought about by differences in both natural and fishing mortality as well as recruitment.

While the changes in the commercial length frequencies for subblocks 13C, 13D, and 13E give an appearance of being only relatively minor, this contrasts markedly with the trend exhibited by both the geometric mean catch rates and the standardized catch rates, which both indicate a strong decline over the period 2007 – 2012 (**Figure 62**).

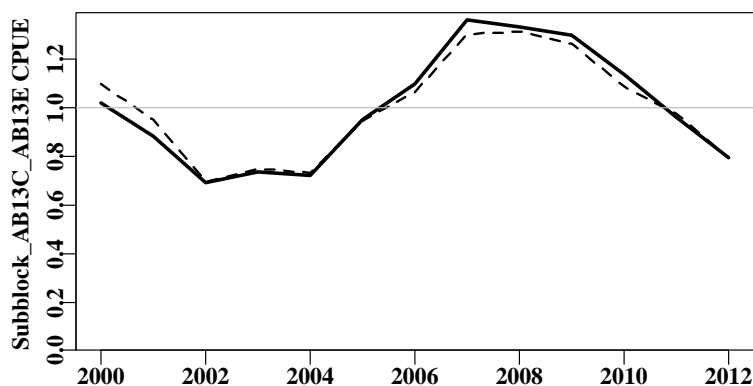


Figure 62. Standardized catch rates for subblocks 13C, 13D, and 13E, over the years 2000 – 2012. The dotted line is the geometric mean while the solid line represents the optimum standardized model.

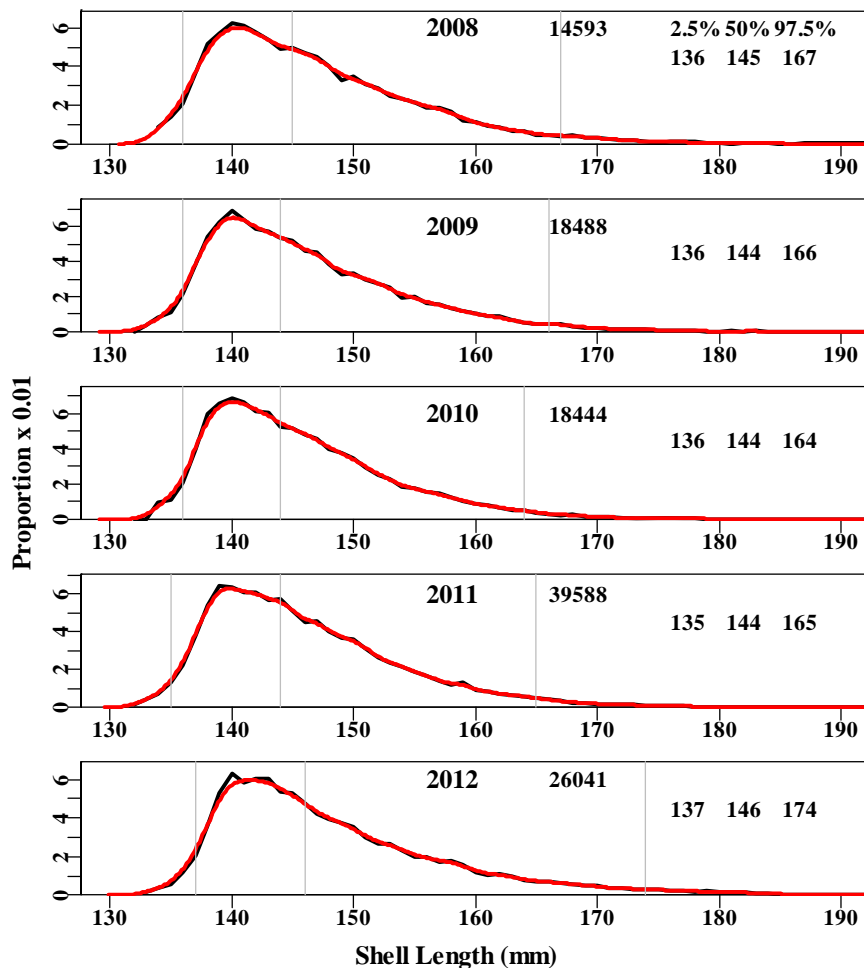


Figure 63. The length frequency of commercial catches from subblock 13C, 13D and 13E each year from 2008 – 2012. The black lines are the observed frequencies while the red lines are the empirical density profile. The number above 170mm is the number of observations each year; the percentages are each year’s quantiles.

17.8.2 Discussion

There is only a minor trend in catch rates in the western blocks over the last five years (2008 – 2012), and this is reflected in the commercial length frequency data, at least from 2008 - 2011. The number of observations is large in each year which increases the confidence in the representativeness of the available data. However, while the visual appearance of the length distribution changes seems only minor the potential effect of the changes that have become apparent can only be determined by fitting a formal stock assessment model to the available data. It is not apparent, simply by making the visual comparison (or even a comparison of quantiles), whether the changes, for example the rightward shift in 2012, is due to a lack of recruitment prior to 2012 forcing the fishery to focus more on larger abalone, or if some other factors is having an influence. A formal model may at least determine whether data from earlier years were consistent with a lack of recruitment in 2012.

It is also possible that this analysis, which combines data from many subblocks, might be obscuring trends. If the outcome from a formal stock assessment model was unclear then a more detailed consideration of smaller groups of western subblocks might provide insight into what is driving stock trends.

When the data for the Actaeon subblocks (13C, 13D, and 13E) are considered it is again the case that no consistent trend is observable between 2008 and 2012 in the length frequency data from commercial catches as measured in the processing sheds. The length frequencies from the years 2008 and 2012 are both somewhat right shifted relative to the other three years (**Figure 61**). As in the west, the changes in length frequencies appear visually to be relatively minor. This is especially the case when they are contrasted with the very strong trend exhibited by the catch rates from 2008 to 2012 (**Figure 62**). Catch rates in 2012 are only 59% of those in 2008, while this degree or direction of change is not apparent in the length frequency data.

It is clear that the proportional distribution of the length frequency of the commercial catch only has limited information about relative abundance. Because it is scaled to the total catch, some measure of how difficult it was to find and catch abalone need to come from other data. Once again, a formal stock assessment model might provide for an interpretation of events in the stock which is not available from the empirical performance measures themselves.

It is also possible that the sampling of commercial length frequency within catches is not representative of all catches taken within the areas characterized. While they are admittedly relatively important catch processors, only two abalone processing factories permit the regular measurement of catches by their staff using the data logging measuring boards. Across the two processors there were 35 divers recording landings. While only 4 divers out of 35 landed to both processors about 40% of processor 1's records and about 26% of processor 2's records came from those shared divers. Despite this, 60% and 75% of records respectively came from divers who only landed to a specific processor. If the proportion of samples in a particular year came from a particular processor and the processors differed to the degree observed, then changes that appear to be a reflection of the fishery would actually be a reflection of the sampling.

This issue emphasizes the need to obtain lengthy time series of length frequencies of catches so that such sources of variation become part of background noise on any signal that derives from how the stocks are changing through time.

18 Model-Based Performance Measures

18.1 Introduction

18.1.1 The Trouble with Models

There are numerous performance measures available for use in the management of fisheries and these can be directly estimated using simple observations collected directly from a fishery (catch rates, spatial distribution of catch, distribution of catches amongst divers, etc). Given this, it ought to be asked why it is necessary to go to the expense and difficulty of developing stock assessment mathematical models, which are only understood by a minority? The simple answer is that while empirical PMs often enable us to describe a fishery and its activities they do not allow us to gain an understanding of the underlying dynamics that are affecting the stock. On the other hand, formal stock assessment models allow the derivation of analytical PMs, including the spawning biomass, B^{Sp} , which can be compared with B_0 the predicted spawning biomass had there been no fishing, which estimates the degree to which the stock or population has declined from its average maximum size. Trying to understand the dynamics of fished stocks and how they are influenced by fishing and management changes requires some way of following the dynamics of the populations making up the fished stock and formal assessment models are the best current approach for doing this.

Because of their high value, abalone fisheries tend to be fished relatively hard and, unfortunately, as a consequence many fisheries around the world have declined badly and some have collapsed (Hobday *et al.*, 2001). Despite this seemingly general pattern with abalone fisheries, in Australia there are a number of fisheries which have continued at productive levels for up to 50 years (Mayfield *et al.*, 2012). However, simulation work conducted in this current project, described in section (20.3.4), has concluded that it is quite possible that abalone fisheries can be slowly depleting to risky levels over long periods, such as fifty years, despite being exploited in a seemingly sustainable manner (Haddon and Helidoniotis, 2013). The current introduction of what is currently assumed to be more risk averse management into South Australia and Tasmania may prevent the determination of which of these conclusions is correct. Nevertheless, the need to understand the dynamics of exploited populations of abalone appears greater than ever.

While the stock assessment modelling of abalone stocks would appear to be an urgent need this does not necessarily mean it is possible to do it in a valid manner. If formal stock assessment models are to work successfully and validly, they require data streams that are informative about the dynamics of the modelled stock. If the available data are too noisy, are not representative, or are not consistently collected through time then there is a possibility that either a stock assessment model will give mistaken and misleading advice or possibly not be able to converge on a biologically reasonable answer.

For many species it is possible to collect data that is sufficient to produce plausible and repeatable results, which, when acted upon by management lead to the expected outcomes and lead to confidence that the fishery dynamics have been reasonably approximated (e.g. Punt *et al.*, 2001). However, with abalone fisheries, because of the spatial heterogeneity of biology between populations (although see Helidoniotis and Haddon,

2013) the question invariably arises as to whether the data that does get collected is ever representative of more than the small areas of a fishery relating directly to wherever the samples came from.

Despite the now classical view that catch rates in abalone fisheries is uninformative, they appear to be useful for monitoring stock dynamics, at least in south-eastern Australia. In addition there are ways of obtaining large samples of the length frequency of the commercial catch (for example, in Tasmania data logging measuring boards are placed in processing sheds and staff there routinely measure thousands of shells). Abalone are very difficult to routinely age and so length-based models have been used in their assessments (Breen and Smith, 2008; Breen *et al.*, 2003; Gorfine *et al.*, 2005; McKenzie and Smith, 2009; Worthington *et al.*, 1998). These models require estimates of growth, size at maturity and related biological details such as weight at length.

Whenever there are alternative model structures available the problem of model selection arises (Burnham & Anderson, 2002). While there are many formal statistical measures of model performance (such as the AIC – Akaike’s Information Criterion or models can be compared using likelihood ratio tests) in addition the biological plausibility of model outcomes should also be considered (Haddon, 2011; Helidoniotis *et al.*, 2011; Helidoniotis and Haddon, 2013).

18.2 Surplus Production Models

18.2.1 Introduction

A comprehensive description of surplus-production modelling is provided in Haddon, 2011) and only a relatively brief treatment of the details required for the modelling used with abalone is included here.

Surplus-production models (S-P models) are one of the simplest analytical methods available that provides for a stock assessment that, if certain assumptions are met, can produce estimates of fishing mortality and of stock biomass through time. Performance measures relating to these two variables are considered to be the most informative about a stock’s status (Quinn and Deriso, 1999).

The ideas behind S-P models were first described in the 1950s (Schaefer, 1954, 1957), but Schaefer’s original analytical strategy involved assumptions requiring equilibrium and the approaches used now are non-equilibrium (Prager, 1994; Haddon, 2011). They are relatively simple to apply, mainly because they pool all aspects of production (the combined effects of recruitment, growth, and mortality) into a single analytical function. The model can be used with either total numbers or total biomass of the fished stock but age- and size-structure, along with sexual and other differences, are ignored. The minimum data requirements to fit such models are time-series of an index of relative abundance and associated catch data. The index of stock abundance is often catch-per-unit-effort (CPUE) but could be some fishery independent abundance index (e.g., from trawl surveys, acoustic surveys) or both could be used.

As a result of age and size being ignored and the fact that, in Australia, CPUE is generally used as an index of relative abundance, S-P models focus on the available exploita-

ble biomass rather than spawning or mature biomass. This distinction is important if the harvest strategy put in place adopts targets and limits relating to spawning biomass rather than exploitable biomass (as in the Commonwealth Harvest Strategy described in DAFF, 2007). The two types of biomass are related, not necessarily in a linear fashion, and are rarely the same, depending on the size at maturity.

To conduct a formal stock assessment it is necessary, somehow, to model the dynamic behaviour of the exploited or spawning stock. The first versions of S-P models assumed that the fishery was always in equilibrium with the fishery being imposed, however, this is a risky assumption as such models can underestimate fishing mortality when the stock is being depleted (Haddon, 2011). Here we will only consider non-equilibrium S-P models. One objective is to describe how the stock has responded to varied fishing pressure. By studying the impacts on a stock of different levels of fishing intensity it is possible to gain information about its productivity. If statistics are collected, the process of fishing a stock can provide information about how the stock responds to perturbations (the extra mortality, above natural mortality, imposed by fishing). If a reduction in the stock size cannot be detected reliably (i.e., catch rates or survey results are hyper-stable relative to stock size), then stock assessment will be difficult, unreliable, or even impossible. A further advantage of non-equilibrium models over equilibrium models is that it becomes possible to determine whether or not the data available has enough information to produce a workable assessment from which conclusions can validly be drawn.

The usefulness of any model is directly related to how representative the available data is for the fished stock and whether the index of relative abundance really does provide a clear index of relative stock size. If the index of relative abundance is informative but a set of ageing data is not necessarily representative then we might expect a surplus production model potentially to be more useful for the provision of management advice than a more advanced age-structured model (Ludwig and Walters, 1985; Punt, 1995). However, while catch rate data may well be representative of a fishery it is often difficult to demonstrate that such data is representative of the state of the underlying stock.

18.2.2 The Requirement for Contrast in CPUE

The S-P models described here are dynamic in that they do not require an assumption of an equilibrium existing between the catch rates and the effort, as was required by the very first such models. Hilborn (1979) analyzed many examples where this assumption was made and demonstrated that the data used were often too homogeneous; they lacked contrast and hence were uninformative about the dynamics of the populations concerned. For the data to lack contrast means that fishing catch and effort information is only available for a limited range of stock abundance levels, which may come about through there being only a short time-series of data available, stable conditions, the fishery being highly depleted, or developing steadily through time with only unidirectional changes in catches and catch rates.

Unfortunately, with more recent versions of S-P models, by removing the requirement for equilibrium it is sometimes, perhaps often, the case that it becomes impossible to obtain a stable fit of such a model to available data. This merely reinforces the fact that such data is uninformative and means some other approach, using either or more data

streams, for assessing the fishery’s status will be required. This is not a disadvantage as it is better to know that an assessment isn’t possible than to use one invalidly.

18.2.3 The Possible Outputs

A non-equilibrium Schaefer model equivalent can produce an output table of predicted exploitable biomass and related statistics (**Table 23**). Using two time blocks relating to LML changes of 127 – 132 from 1985 – 1989, which included a change to 132 in 1987, and 1990 – 2011 during which the LML was 140 mm, this gave two estimates of catchability (although in this case there is almost no difference between them 1985 – 1989 being 0.02501 and 1990 – 2011 being 0.02497), and in an age- or size-structured model it might be better to use different selectivity estimates rather than different catchability estimates.

Table 23. The inputs and base outputs from a surplus production model. The inputs are the catch and CPUE columns, which relate to blocks 9 – 12 on the west coast. The outputs are a time series of predicted exploitable biomass, derived from the model itself, equ (9), and the predicted CPUE, derived from the predicted biomass and the closed form estimates of catchability, that is combining equations (10) and (11).

Year	Catch	CPUE	PredictedB	Pred_CPUE
1985	1018.884	73.565	3278.317	81.999
1986	742.347	78.223	3072.464	76.850
1987	868.023	78.311	3122.057	77.968
1988	715.104	79.805	3051.422	76.204
1989	585.651	82.151	3125.876	78.063
1990	532.214	82.990	3338.023	83.361
1991	566.507	88.245	3624.207	90.508
1992	611.126	91.477	3897.130	97.324
1993	548.256	100.152	4138.264	103.346
1994	499.406	106.578	4447.732	111.074
1995	478.919	120.656	4804.990	119.996
1996	427.787	131.610	5170.227	129.117
1997	657.496	143.157	5561.311	138.884
1998	480.660	148.531	5681.598	141.888
1999	645.049	156.693	5963.169	148.920
2000	960.741	156.464	6038.589	150.803
2001	940.434	143.368	5785.855	144.492
2002	911.717	140.285	5593.065	139.677
2003	954.589	137.921	5455.159	136.233
2004	935.142	128.719	5290.938	132.132
2005	940.138	124.036	5163.530	128.950
2006	951.849	125.112	5042.840	125.936
2007	1030.043	122.266	4920.116	122.871
2008	984.599	111.036	4727.623	118.064
2009	835.676	121.224	4590.915	114.650
2010	839.659	117.584	4608.342	115.085
2011	832.946	118.142	4621.220	115.407

The model fit is visually very good (**Figure 64**).

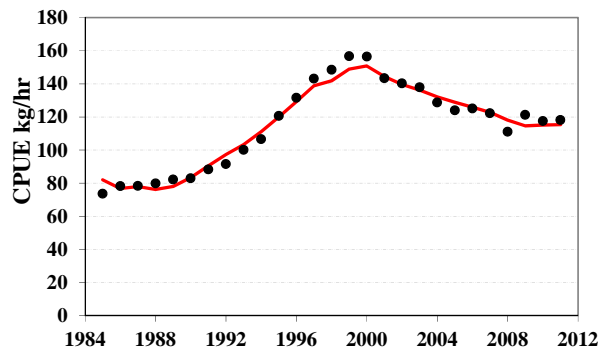


Figure 64. Observed catch rates, as black dots, against the predicted catch rates, red line, from the surplus production model inputs and outputs listed in **Table 23**.

This suggests that the stock was depleted to about 38.5% of the unfished exploitable biomass in 1985, but was recovered up to about 54.3% in 1999 (**Table 24**); however, because the LML had changed markedly between 1985 (127mm) and 2011 (140mm) while these two numbers both relate to the K parameter, the exploitable biomass they refer to was undoubtedly changed by increasing the LML. This is one reason why using spawning biomass as a performance measure is less prone to confusion as it is not influenced by changes in LML.

Table 24. The model input parameters used to fit the model to the data (r,K, B0, p) and the closed form parameters, q1, q2, plus the derived statistics of management interest.

Parameter	Value	Derived Statistic	Value
r	0.4033	B2011	4621.220
K	8513.696	B2011/K	0.5428
B0	3278.317	MSY	858.390
p	1	B0/K	0.3851
q1	0.025013	SSQ	0.041777
q2	0.024973	LL	49.05048

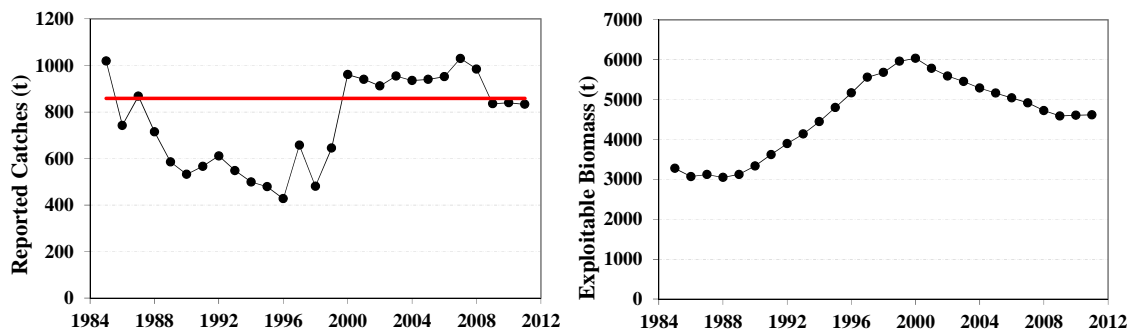


Figure 65. The left hand graph illustrates the observed catches, black line, and the MSY, red line, while the right hand graph illustrates trends in the exploitable biomass in blocks 9 – 12.

By chance the MSY of 858 t is similar to the catches currently being taken from blocks 9 – 12, and, in this case, because $p = 1.0$, the production curve is symmetric so the biomass that should generate the MSY is $K/2$, which is 4255 t, which is slightly lower than that estimated for 2011. The conclusion from the model, therefore, is that the current harvest rate should be sustainable assuming recruitment is maintained at current levels. However, the MSY is not necessarily the best target to have for a fishery. The profitability of fishing relates to both the amount of catch taken but also to the effort required to catch that take. By allowing the stock to rebuild to a higher level and taking less than the MSY the catch rates should increase. There is a balance within this trade-off between catch level and consequent catch rates where profitability is maximized and that provides a more economically efficient target to manage the fishery towards. Unfortunately, arrangements are more complex than that within Tasmania. Being a quota fishery there are many divers who do not own quota but each year fish other people's quota on a leasing system where they are paid a rate for each kg landed and any remainder goes to the quota owner. If the amount paid per kg fails to increase when fishing costs increase, as often happens, then it is the diver who faces the financial risk of catching the abalone. This has led to quota owners not necessarily having the same agenda as the divers. In Tasmania it sometimes becomes apparent that quota owners aim to maximize catches while divers would prefer to maximize catch rates. Thus it is possible in this fishery for some sectors to avoid the disadvantages of fishing to the MSY. These conflicting objectives between the two sectors makes management more complex.

18.2.4 Surplus Production Model Equations

A deterministic non-equilibrium surplus production model was used to relate CPUE to stock biomass, which assumes that CPUE are an indicator of relative stock abundance. A modified Schaefer model, the Pella Tomlinson model (Pella and Tomlinson, 1969) was used to determine the exploitable biomass (tonnes) at time $t+1$ using catch and CPUE data from time t (the time step is in years).

$$B_{t+1} = B_t + \left(\frac{r}{p}\right) \times B_t \times \left(1 - \left(\frac{B_t}{B_0}\right)^p\right) - C_t \quad (9)$$

where B_{t+1} is the exploitable biomass at the end of year t or at the beginning of year $t+1$, B_t is the exploitable biomass at the start of year t , r is the growth rate in population abundance (derived from the intrinsic rate of natural increase), B_0 is the virgin Biomass or the average biomass prior to exploitation (derived from the idea of carrying capacity), and p is the asymmetry parameter. If $p = 1$ then the production curve would be symmetric, which means the model simplifies to the dynamic Schaefer model (Haddon, 2011).

The predicted CPUE is linked to the deterministic time series of biomass estimates using the following equation:

$$\hat{I}_t = \frac{\hat{C}_t}{E_t} = q\hat{B}_t^\lambda e^\varepsilon \quad (10)$$

Where I_t is an index of relative abundance for year t , q = catchability coefficient, λ is the production curve asymmetry parameter, and e^ε represents multiplicative log-normal residual errors with a constant variance (i.e where $\varepsilon = N(0;\sigma^2)$). A closed form estimate of the catchability q is given by the geometric mean of the time series of individual q estimates (observed CPUE / predicted biomass)

$$\hat{q} = e^{\frac{1}{n} \sum \text{Ln} \left(\frac{I_t}{\hat{B}_t} \right)} \quad (11)$$

The introduction of a change in the Legal Minimum Length would immediately alter the amount of exploitable biomass and hence its relationship with CPUE. It could be argued that altering the LML simply reduces the amount of exploitable biomass and doesn't affect the catchability. However, being a diver fishery where there is a direct interaction between the divers and the taking of abalone, changes to the LML are likely to influence diver behaviour and thereby influence the dynamics of fishing. The issue remains debatable and is difficult to test. To account for such management induced changes (which cannot be included in the standardization), a separate q estimate can be estimated for each period having a different LML. Surplus production models do not use selectivity in their dynamics so altering the possible catchability is the only option left to explore.

It is necessary to log transform both the predicted and observed CPUE to normalise the residual errors. The model is fitted to the data using maximum likelihood methods

$$LL = \frac{1}{\sqrt{2\pi \hat{\sigma}^2}} \prod_t e^{-\frac{(\text{Ln} I_t - \text{Ln} \hat{I}_t)^2}{2 \hat{\sigma}^2}} \quad (12)$$

Where the product is over all years (t) for which CPUE data is available and where

$$\hat{\sigma}^2 = \sum_t \frac{(\text{Ln} I_t - \text{Ln} \hat{I}_t)^2}{n} \quad (13)$$

and n is the number of observations. Equation (12) can be simplified (Haddon, 2011) to become:

$$LL = -\frac{n}{2} [\text{Ln}(2\pi) + 2\text{Ln}(\hat{\sigma}) + 1] \quad (14)$$

Given a times series of catches (C_t) and starting parameter estimates the model produces a series of expected biomass values \hat{B}_t . Given the catchability coefficient q , Eq(11), the predicted biomass estimates \hat{B}_t are used to produce a series of expected CPUE using eq (10). The initial biomass B_0 is estimated directly as a separate parameter (Haddon,

1998). Statistics of interest, which relate to when $p = 1.0$, include the maximum sustainable yield:

$$MSY = \frac{rK}{(p+1)^{\frac{(p+1)}{p}}} \quad (15)$$

Which, if $p=1$ simplifies to $rK/4$.

The fishing mortality rate that should lead to the MSY:

$$F_{MSY} = qE_{MSY} = q \frac{r}{2q} = \frac{r}{2} \quad (16)$$

And the current state of depletion:

$$B_{\%}^E = \frac{B_{2011}^E}{K} \quad (17)$$

18.2.5 Bootstrapping to Characterise Uncertainty

To characterize the uncertainty surrounding each of the model outputs the log-normal CPUE residuals are bootstrapped and combined with the optimum model fit to generate bootstrap CPUE samples to which the model was refitted.

$$I_t^* = \hat{I}_t \times \left(\frac{I_t}{\hat{I}_t} \right)^* \quad (18)$$

where I_t^* is a bootstrap CPUE sample for year t , \hat{I}_t is the optimum model fit in year t to the original CPUE data, and I_t is the observed CPUE in year t . 1,000 bootstraps will provide estimates of the 10% 50% and 90% percentiles from the predicted results and permit a representation of the uncertainty around all model outputs. Similarly, the maximum sustainable yield (MSY) and model parameters can be estimated for comparison. Alternative methods for characterizing uncertainty, including the use of likelihood profiles and Bayesian posteriors are described, using abalone as an example, in Haddon (2011).

18.2.6 Model Projections

Once an optimal model fit had been achieved and the bootstrapping completed the model dynamics in each bootstrap sample can be projected forwards in a deterministic fashion for possibly 10 years under different assumed TAC levels. A minimum of 1000 projections should be made to provide for a range of outcomes and the 10% 50% and 90% percentiles can be estimated from the predicted results. By plotting the projections and the central 90% spread of predicted values for the projected dynamics the implications of applying alternative TACs can be made visually apparent. In this way the risks of alternative management options can be explored, aided by the diagrams and equivalent tables of probabilities for different outcomes.

18.2.7 The Importance of Management Objectives

Of course, before any particular outcomes and their related TACs can be selected, to make such decisions defensible it is best to have an explicitly stated objective towards which the fishery should be managed. Thus, one could select a given catch rate as a target that the fishery aspires to achieve and maintain, or a given state of stock depletion (ideally one known to be productive of an acceptable size of harvest). Whatever the decision made as to a fisheries objective, if it is made explicitly then the particulars of various management decisions become straightforward to defend.

18.3 Size-Based Models

18.3.1 The Data Requirements

Since 1990, the major non-equilibrium or dynamic modelling option for abalone fisheries has been size or length-based modelling (Sullivan et al., 1990; Sullivan, 1992; and a fully developed model for abalone is described by Breen et al. (2003), which was similar to that produced earlier for rock lobster (Punt and Kennedy, 1997).

Such models have been extended slightly in Tasmania to enable the inclusion of the emergence from crypsis of under-sized abalone to enter the fishery (the equations are given below).

These size-based models can also be called integrated analyses because they are able to integrate numerous different data streams including such things as CPUE, catches, tagging data, length-frequency of catch data, and an array of biological properties. The biological properties required include a detailed description of growth that comprises the mean growth increments for each time step and for each size class, plus the expected spread about those mean increments; being size-based the growth description is critical to the dynamics. The standard way of including the description of growth is by generating a growth transition matrix which characterizes the relative probabilities of growing from a given size class of animals into the expected spread of other size classes.

In addition to growth an estimate of size at maturity is required, if details of spawning biomass are wanted (which is usual), and estimates of natural mortality or of survivorship are required. If emergence is to be included in the dynamics then estimates of the emergence at size are required. Including emergence can have value because all recruitment tends to be into cryptic habitats and all fishing mortality is on emergent animals; however, emergence is not a necessary requirement.

Decisions also have to be made about how the model is to be initiated. If the complete catch history of a fishery is known then the modelling can begin with the fishery in unfished equilibrium, however, if the early history is highly uncertain, which is often the case and certainly the case in Tasmania, then some method for initiating the size structure of the modelled population at the start of the fitted time series needs to be used.

18.3.2 The Possible Outputs

The possible outcomes from size-based models can be more numerous and more detailed than with surplus production models. The same outcomes as from the surplus production models can be derived but in addition to those the expected size distribution of both the catch and the population in any year can be estimated, the selectivity of the fishing, and the size-based fishing mortality. More generally, using such models permits the estimation of the proportion of the spawning biomass protected by a given LML and the annual recruitment levels to be estimated. Many more details concerning the dynamics of the stock can be discerned.

Generally the management options within Australian abalone fisheries relate either to the TAC, the LML, or some form of spatial management. The effects of altering such things as the TAC and/or the LML can both be estimated in model projections for size-

structured models. This would provide for more management options than the surplus production modelling and many more options than can be obtained from the use of empirical PMs and their related control rules.

Typical output from a size based model will include the predicted catch rates, the annual exploitable and mature biomass, the harvest rate (proportion of exploitable biomass taken by fishing), the relative recruitment levels each year, and the depletion levels of the two types of biomass (**Table 25**).

Table 25. Typical output from a size-based model, as described by the equations below. Input data included catches and CPUE from 1985 – 2010 and length frequency of commercial catches from 2000 – 2010. PredCE is the predicted catch rate, CEResid is the catch rate residuals, ExpltBt is the exploitable biomass, MatBt is the mature biomass, Deplet is the spawning biomass depletion level while DepletEB is the depletion level of the exploitable biomass.

Year	CPUE	PredCE	Catch (t)	CEResid	ExpltBt	MatBt	Harvest	Recruit	Deplet	DepletEB
1985	0.639	0.649	987.291	-0.010	2696.895	3408.589	0.366	0.823	0.317	0.270
1986	0.679	0.669	712.706	0.010	2780.102	3488.758	0.256	0.839	0.324	0.278
1987	0.680	0.675	855.502	0.005	2962.643	3751.539	0.289	0.808	0.348	0.301
1988	0.693	0.691	690.051	0.002	3033.532	3756.403	0.227	0.789	0.349	0.308
1989	0.714	0.721	553.570	-0.007	3162.093	3871.064	0.175	0.956	0.360	0.321
1990	0.721	0.715	516.445	0.006	2949.610	4067.223	0.175	0.938	0.378	0.318
1991	0.767	0.765	552.672	0.001	3157.435	4329.197	0.175	1.086	0.402	0.340
1992	0.795	0.808	610.114	-0.013	3331.455	4627.736	0.183	1.074	0.430	0.359
1993	0.870	0.863	548.073	0.007	3558.218	4927.989	0.154	1.071	0.458	0.383
1994	0.926	0.937	498.230	-0.012	3866.807	5347.667	0.129	1.102	0.497	0.417
1995	1.048	1.042	478.792	0.006	4300.233	5793.831	0.111	1.071	0.538	0.463
1996	1.143	1.144	427.766	-0.001	4719.396	6222.137	0.091	0.969	0.578	0.508
1997	1.244	1.245	657.496	-0.002	5138.087	6649.880	0.128	0.907	0.618	0.553
1998	1.290	1.288	480.660	0.002	5314.249	6763.078	0.090	1.042	0.628	0.572
1999	1.361	1.353	645.049	0.008	5580.108	6919.184	0.116	1.051	0.643	0.601
2000	1.359	1.348	960.741	0.011	5560.883	6889.605	0.173	0.919	0.640	0.599
2001	1.245	1.264	940.434	-0.019	5216.315	6643.974	0.180	0.970	0.617	0.562
2002	1.219	1.218	911.717	0.001	5023.449	6426.483	0.181	1.118	0.597	0.541
2003	1.198	1.184	954.589	0.014	4884.705	6208.746	0.195	0.994	0.577	0.526
2004	1.118	1.124	935.142	-0.006	4637.788	6040.571	0.202	0.878	0.561	0.500
2005	1.077	1.088	940.138	-0.010	4487.334	5962.089	0.210	1.312	0.554	0.483
2006	1.087	1.079	951.849	0.008	4450.756	5809.525	0.214	0.958	0.540	0.479
2007	1.062	1.046	1030.043	0.016	4316.731	5738.654	0.239	0.965	0.533	0.465
2008	0.965	0.997	984.599	-0.033	4113.801	5715.904	0.239	1.000	0.531	0.443
2009	1.053	1.023	835.676	0.030	4218.946	5611.773	0.198	1.000	0.521	0.454
2010	1.021	1.031	835.755	-0.009	4252.633	5611.386	0.197	1.000	0.521	0.458

To fit the model in the example to the available data used two different sources of likelihood and a likelihood penalty (**Table 26**) imposed on recruitment variation (to restrict recruitment variation to plausible levels).

The dynamics of the stock can be inferred from the model outputs (**Figure 66**). The age of animals as they enter the fishery at 140 mm is approximately 6 years, with modal age classes remaining discernible up to the age of five with age six animals beginning to merge with the larger, older animals (**Figure 66**).

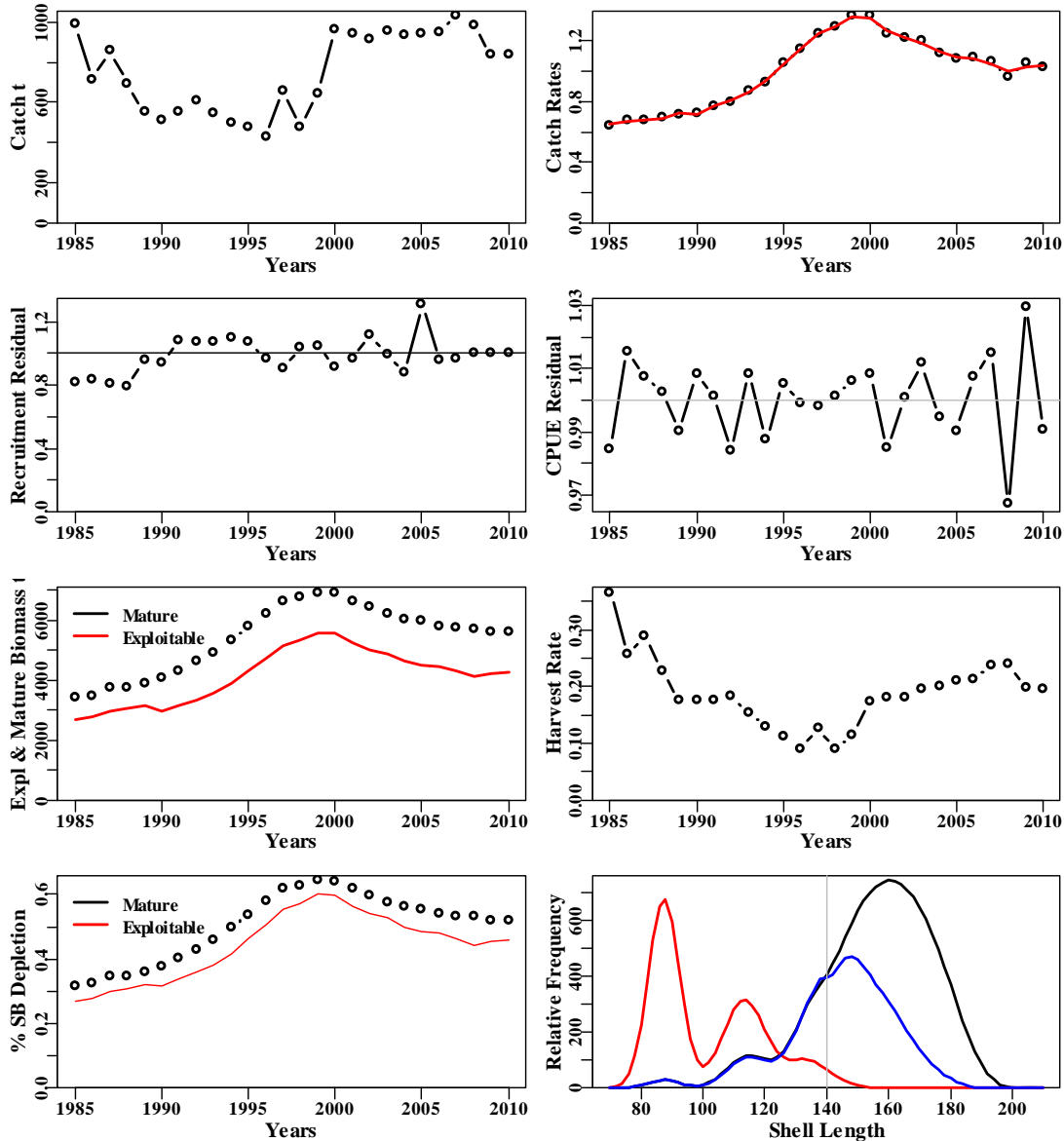


Figure 66. Graphical representations of the various outputs from the size-based model (**Table 25**). The fit to the catch rates appears good and the distribution of residual is illustrated in the 2nd row right column. A relatively strong recruitment is implied as occurring in 2005 and both 2008 and 2009 appear a little unusual in terms of cpue residual. A kink in the exploitable biomass in 1990 was produced by changing the LML from 132 to 140, although the change from 127 to 132 in 1987 is not apparent. The exploitation rate attained a minimum from 1996 – 1998 in the years before zonation was introduced. The current depletion level is apparent in the predicted length frequency of the emergent animals; the black line in the bottom right graph represents the unfished state and the blue line the current state. The red line represents the cryptic animals starting at age 4 animals in the 87mm mode.

Table 26. Likelihood components from fitting the model to the input data giving equal weight to the catch rate and length frequency data.

	Likelihood	Weighting
Catch Rates	-76.751	1
Length Frequency	285.000	1
Recruitment Penalty	13.757	
Total Log-Likelihood	222.006	

The fit to the commercial catch length frequency data varied between years but was often better the larger the number of observations there were (**Figure 67**).

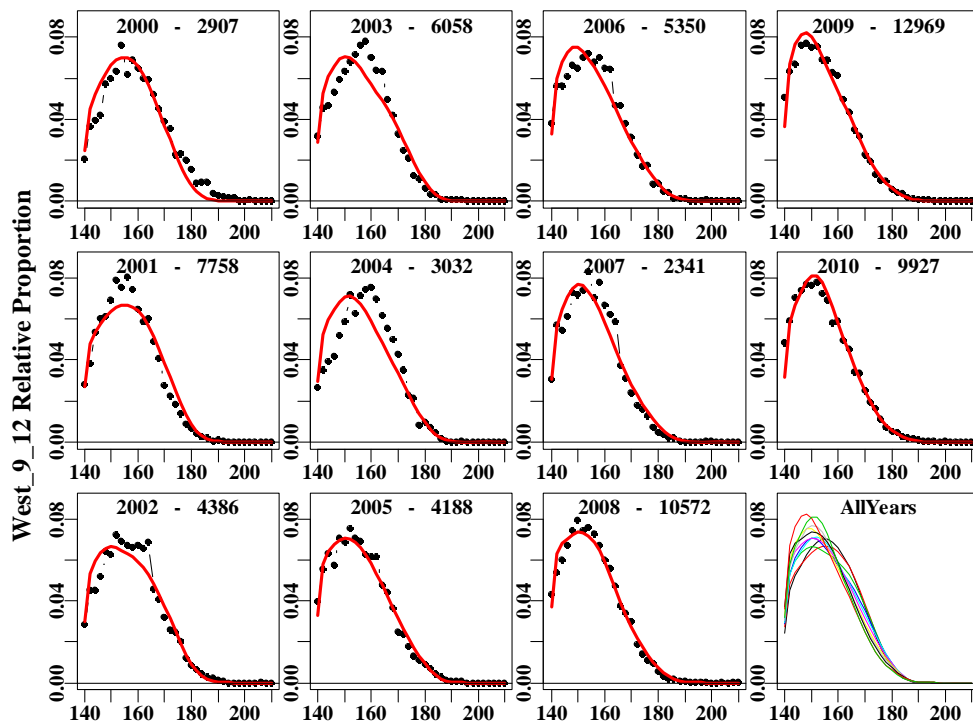


Figure 67. The commercial catch length frequency data and the respective fits (red lines) from the size-based model. The data from 2008 onwards was collected using electronic data logging measuring boards. The graph at bottom right illustrates the variation expected between years.

18.3.2.1 Projections

By simulating the variation around potential recruitment by mimicking previously estimated recruitment levels and variation, it is possible to project the current state forward (**Figure 68**).

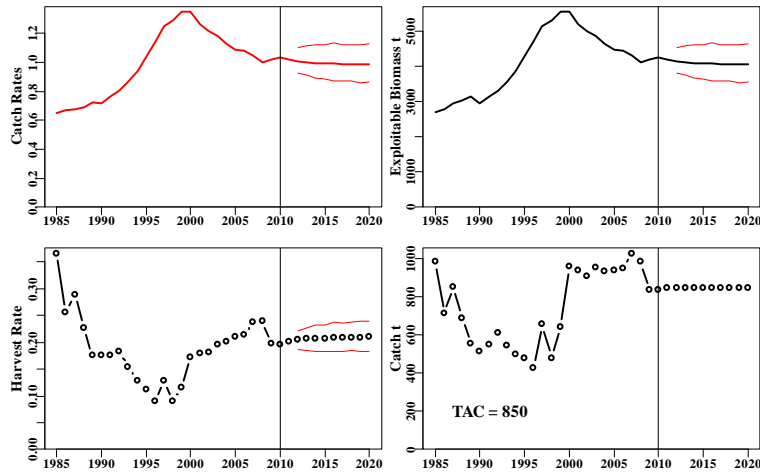


Figure 68. Projections from 2010 based on a TAC of 850 t and generating random recruitment time series that mimic previously estimated recruitment. The fine red lines are the inner 95% of all projections while the central projected line is the median.

By running such projections forward under different levels of TAC or with different LML, which can be simulated by manually altering the selectivity used in the dynamics, the relative likelihood of achieving different outcomes can be determined. The types of outcome can include having a specific spawning biomass depletion level, or a particular harvest rate, or any other measurable statistic. In this way, if, for example, rebuilding of a stock's biomass were required, it would be possible to make predictions as to how long such rebuilding might be expected to take; as long as conditions during the rebuilding period remained similar to conditions that occurred during the period of the known dynamics.

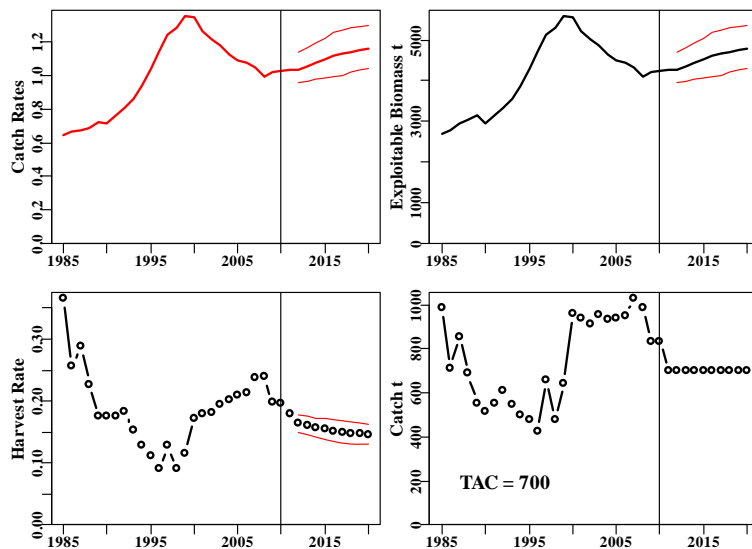


Figure 69. Projections from 2010 based on a TAC of 700 t and generating random recruitment time series that mimic previously estimated recruitment. The fine red lines are the inner 95% of all projections while the central projected line is the median.

Projecting forward with a TAC of 850 t predicts stability 50% of the time, but that also means there is also an equal chance of the stock either declining or rising. If there is a

wish to rebuild biomass and increase catch rates then a TAC of 700 t is predicted to reverse the downward trend in catch rates exhibited since 2000, although the rate of increase would be expected to be lower than the decrease seen over the last 10 years.

18.3.3 The Importance of Structural Assumptions

The model equations that represent the dynamics can take a number of forms and each implies a slightly different order of events. In the version used, described below, the order of events within each year is assumed to be as follows, assuming that N_t represents the number of abalone at the very start of a year, the algorithm used was:

1. The numbers at size in crypsis at time t , N_t , are reduced by those which are going to emerge in that year ($N^C - EN^C$);
2. The remainder undergo half of the natural mortality that occurs in crypsis, which need not be the same as for emergent abalone $C_S(N^C - EN^C)$;
3. The remainder then grow through application of the growth transition matrix ($GC_S(N^C - EN^C)$);
4. These then undergo the remaining half of natural mortality and any recruitment occurs to produce the estimate of numbers of abalone in crypsis at the end of the year (or start of the next) at time $t+1$ [$C_S(GC_S(N^C - EN^C))$] + R.
5. The emergent animals from crypsis are added to the numbers at size for emergent abalone ($N^E - EN^C$);
6. The remainder then undergo half the natural mortality that occurs on emergent abalone followed by growth ($GO_S(N^E - EN^C)$);
7. Finally, the remaining half of natural mortality is applied to generate the estimate of abalone in crypsis at the start of the next year, or at time $t+1$ [$O_S(GO_S(N^E - EN^C))$].

Alternative structures might have the natural mortality being applied in one go, or the order of growth, mortality, and emergence being different. Such changes do not tend to make much difference to the overall trends expressed across years but whatever sequence of dynamics are adopted as best approximating the dynamics of the fishery being analyzed, these dynamics have then to be applied consistently in all stages of the modelling else there can be internal inconsistencies leading to biases in estimates of such statistics as the fishing mortality rate (if fishing is mistakenly applied before a population is grown, for example, there will be less exploitable biomass available to be fished and hence for the same catch the fishing mortality will appear higher).

18.3.4 When are Size-Based Models Appropriate?

The data and information requirements of size-based models are significantly greater than for the simpler surplus production models. Surplus production models are only appropriate when there is contrast in the catch and effort data, meaning there is data available from a wide range of stock sizes and of different catch levels. In addition, it is necessary that there exists a relatively simple relationship between catch rates and the amounts of exploitable biomass available to the fishery. These requirements are also necessary for the application of size-based models to be appropriate and these mainly revolve around the available data being representative of the assessed stocks. However, there are other requirements that apply only to size-based models. The growth description that is used needs to be representative, which can be difficult to achieve where the growth patterns appear to be highly variable.

The growth models used relate to descriptions of the expected growth increments for different size classes. These are best determined using tagging data although a question remains whether the tagging process influences the growth expressed or perceived. In the case of the size based model used in blocks 9 – 12 to produce the description of growth used data from three sites across the four blocks were available and in fact the outcomes from analyses indicated there were no significant differences between the growth descriptions expressed in these three sites. Nevertheless, this estimate of growth appears to have been an under-estimate as if it was treated simply as an input to the model the predicted unfished size distribution only matched the size distribution of the catch in a partly depleted stock; which is not really possible. Instead the growth data were used to obtain the shape of the growth curve (the relationship between the three parameters and then the final value of two of the growth parameters (L50 and L95) were allowed to be fitted in the size-based model, which used both the tagging data and the commercial catch at length data to determine more likely parameter estimates. However, the change to the growth curve was then a poor reflection of the tagging data (**Figure 70**). The importance of growth to the dynamics is so great that this is an area that requires further exploration before full confidence can be attributed to these size-based models.

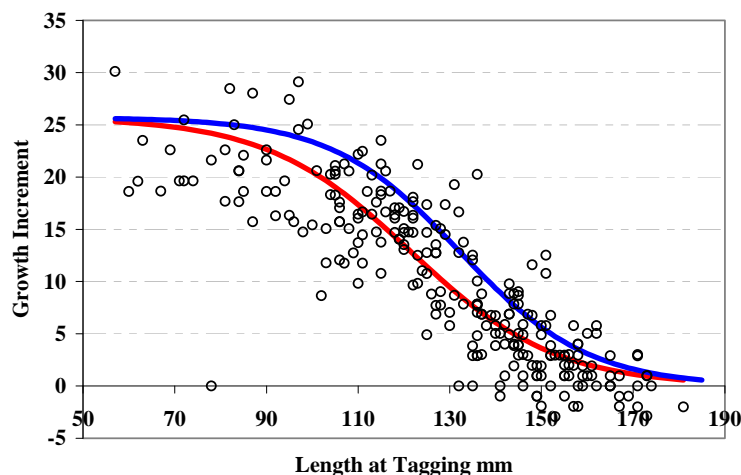


Figure 70. Comparison of the optimal fit to the tagging data (the red line) and the growth required by the assessment model to allow for the catches of larger abalone (blue line). This is Figure 10 from Haddon (2009). The dots are the tagging data from the west coast blocks 9 – 12.

Of great importance is the use of the λ parameter in the relationship between catch rates and exploitable biomass, as in equations (10) and (38). By setting λ to 1.0 this implies that the relationship is linear, which in turn implies that catch rates are directly related to exploitable biomass and that hyper-stability of catch rates is not an issue. While the empirical exploration of CPUE in Tasmania indicates a strong relationship between the two there it should also be examined elsewhere to see if this assumption holds.

All of these assumptions and options remain open to further exploration.

18.3.5 The Model Equations

18.3.5.1 Model Variables

Bolded terms are matrices or vectors. Subscript t relates to year, subscript L relates to length class, while superscript Sp or Ex relates to spawning or mature and exploitable biomasses. Superscript E and C relate to emergent and cryptic components respectively.

- O_s** is a square zero matrix ($n \times n$) with the survivorships, $e^{-(M/2)}$, by size class down the diagonal elements for emergent abalone, this involves the survivorship from half the natural mortality,
- C_s** is a square zero matrix ($n \times n$) with the survivorships, $e^{-(M/2)}$, by size class down the diagonal elements for cryptic abalone, this involves the survivorship from half the natural mortality (which need not be the same as for emergent abalone),
- M** is natural mortality, which can be different in cypsis and emergent populations,
- N_t^E** is a vector of numbers-at-size in year t for emergent abalone, with n size classes,
- G** is a square growth transition matrix ($n \times n$), the same for both sexes,
- E** is a square zero matrix ($n \times n$) with the proportion emergent by size class, Eq (25), arranged along the diagonal elements,
- N_t^C** is a vector of numbers-at-size in year t for cryptic abalone, with n size classes,
- I** is the unit matrix,
- R** is a vector (n) of recruitment numbers (generally zero except for the smallest size classes),
- N^{C*}** is the equilibrium initial population size structure for cryptic animals,
- N^{E*}** is the equilibrium initial population size structure for emergent animals,
- A** is the complement of an annual harvest rate (via a selectivity curve, s) applied to the emergent animals, (**$I-sH$**)
- H** Annual harvest rate,
- s_L** selectivity of length class l ,
- L_{E50}** logistic parameter for the emergence curve, depicts the length at which 50% of cryptic animals become emergent,
- L_{E95}** logistic parameter for the emergence curve, depicts the length at which 95% of cryptic animals become emergent,
- LW** is the class width in mm,
- $\bar{L}_{t,j}$** is the expected mean length of animals starting in size class j ,
- L_{min}** is the minimum size class considered,
- L_{max}** is the maximum size class considered,
- σ^j** is the expected standard deviation for length class j ,
- L_{m50}** logistic parameter for the growth curve, depicts the length at which the growth increment is 50% of the maximum,
- L_{m95}** logistic parameter for the growth curve, depicts the length at which the growth increment is 5% of the maximum,
- $Max\Delta L$** is the maximum growth increment for the inverse logistic curve describing abalone growth, The point at which variation is 5% of the maximum is set at 210mm for the west coast and the 50% point is set at L_{m95} ,

$\text{Max}\sigma_L$	is the maximum standard deviation describing the variation around the mean expected growth increment,
W_L	is the weight in grams of abalone of length L ,
a, b	are the weight at length parameters,
m_L	maturity at length L ,
α, β	are the maturity at length logistic parameters, α/β depicts the length at 50% maturity and $2.\text{Ln}(3)/\beta$ is the inter-quartile distance,
L_{s50}	logistic parameter for the selectivity curve, depicts the length at which 50% selection occurs,
L_{s95}	logistic parameter for the selectivity curve, depicts the length at which 95% selection occurs,
σ_R^2	is the variance of the recruitment residuals,
q_p	is the catchability of period p (either 1985-1989 or 1990- 2007),
I_t	is the standardized catch rate in year t .
$WtCE$	is the weight given to the catch effort contribution to the negative log-likelihood,
$WtLF$	is the weight given to the proportion length frequency data to the negative log-likelihood,
$WtRec$	is the weight given to the penalty on recruitment variation,
K_t	is the square root of the number of observation of length frequency in each year t .

18.3.5.2 Model Structure

The mortality schedules can differ between the cryptic and emergent population components especially if a constant initial fishing mortality is applied to the emergent population and exactly how the fishing mortality is implemented in the model needs to be reflected in the equilibrium and dynamics equations. The model structure adopted has half of natural mortality occurring followed by growth and fishing mortality, followed by the remaining natural mortality. If natural mortality is implemented as half natural mortality, that is $\mathbf{C}_s = e^{-M/2}$ for cryptic and $\mathbf{O}_s = e^{-(M/2)}$ for emergent, twice in the year, with other dynamics between the natural mortality events then the dynamics can be represented as:

$$\mathbf{N}_{t+1}^C = \mathbf{C}_s \left[\mathbf{GC}_s (\mathbf{N}_t^C - \mathbf{EN}_t^C) \right] + \mathbf{R} \quad (19)$$

and

$$\mathbf{N}_{t+1}^E = \mathbf{O}_s \left[\mathbf{GO}_s (\mathbf{N}_t^E + \mathbf{EN}_t^C) \right] \quad (20)$$

at equilibrium

$$\mathbf{N}^{C*} = \left(\mathbf{I} - \left[\mathbf{C}_s \mathbf{GC}_s (\mathbf{I} - \mathbf{E}) \right] \right)^{-1} \mathbf{R} \quad (21)$$

Consequently, for emergent abalone:

$$\mathbf{N}^{E*} = \left(\mathbf{I} - \mathbf{O}_s \mathbf{GO}_s \right)^{-1} \mathbf{O}_s \mathbf{GO}_s \mathbf{EN}^{C*} \quad (22)$$

If there is an initial estimated fishing mortality rate, this can be defined as the complement of an annual harvest rate and is distributed down the diagonal of an otherwise zero matrix \mathbf{A} :

$$A_L = (1 - s_{L,t}H_t) \quad (23)$$

where A_L is the survivorship of length class L , $s_{L,t}$ is the selectivity of length class L in year t , and H_t is the fully selected harvest rate in year t . With an initial fishing mortality rate there would be no change to the equilibrium for the cryptic component, Eq (21), but the equilibrium numbers for the emergent population would become:

$$\mathbf{N}^{\mathbf{E}^*} = \left((\mathbf{I} - \mathbf{O}_s \mathbf{A} \mathbf{G} \mathbf{O}_s) \right)^{-1} \left(\mathbf{O}_s \mathbf{A} \mathbf{G} \mathbf{O}_s \mathbf{E} \mathbf{N}^{\mathbf{C}^*} \right) \quad (24)$$

Transfer from crypsis into emergence is described using a standard logistic equation (Haddon, 2011):

$$E_L = \frac{1}{1 + e^{-Ln(19)(L-L_E50)/(L_E95-L_E50)}} \quad (25)$$

Where E is the proportion of size class L that are emergent, and $L50$ and $L95$ are the usual logistic parameters defining the lengths at which 50% and 95% are emergent.

The weight at size, W_L , relationship

$$W_L = aL^b \quad (26)$$

Maturity at size, m_L ,

$$m_L = \frac{e^{(\alpha+\beta L)}}{1 + e^{(\alpha+\beta L)}} \quad (27)$$

The elements of the growth transition matrix are defined by:

$$G_{i,j} = \int_{-\infty}^{L_i + \frac{LW}{2}} \frac{1}{\sqrt{2\pi}\sigma^j} e^{-\left[\frac{(L_i - \bar{L}_{i,j})^2}{2(\sigma^j)^2}\right]} dL \quad \left. \begin{array}{l} L_i = L_{Min} \\ L_{Min} < L_i \leq L_{Max} \end{array} \right\} \quad (28)$$

to ensure that all columns sum to 1.0 and to make L_{Max} a plus group the final row of the matrix is modified for each column j as:

$$G_{L_{Max},j} = G_{L_{Max},j} + \left(1 - \sum_{i=L_1}^{L_{Max}} G_{i,j} \right) \quad (29)$$

The expected mean size for each initial size class j is defined using an inverse logistic growth curve that has been found to describe blacklip abalone growth well (Haddon *et al.* 2008):

$$\bar{L}_{i,j} = L_j + \frac{Max\Delta L}{1 + e^{\frac{Ln(19)(L_j - L_m 50)}{(L_m 95 - L_m 50)}}} + \varepsilon_{L_j} \quad (30)$$

Variation around the mean expected growth increment is assumed to be normal with a standard deviation that varies with the growth increment (Haddon *et al.* 2008):

$$\sigma_{L_j} = \frac{Max\sigma_L}{1 + e^{\frac{Ln(19)(L_j - L_m 95)}{(210 - L_m 95)}}} \quad (31)$$

Selectivity for length L in year t is defined as:

$$s_{L,t} = \frac{1}{1 + e^{-\frac{Ln(19)(L - L_s 50)}{(L_s 95 - L_s 50)}}} \quad (32)$$

and because of changes to the legal minimum length (LML), selectivity is defined separately for 1985 – 1986, 1987 – 1989, and 1990 – 2008 (**Table 27**).

Recruitment is distributed between the first two size classes (60-62, and 62-64mm) in a 0.9:0.1 ratio, all other size classes being set to zero. Given an array of recruitment residuals and an average recruitment of \bar{R} , the recruitment levels in each year R_t are given by:

$$R_t = \bar{R} \cdot e^{N(0, \sigma_t^2)} \quad (33)$$

The model estimates the $N(0, \sigma_t^2)$ for each year t .

The model is conditioned on catches, that is, the removals are determined from the catches divided by the exploitable biomass:

$$H_t = C_t / B_t^E \quad (34)$$

Where the exploitable biomass is estimated after half of natural mortality, growth, and emergence (divisible by 1,000,000 to estimate as tonnes):

$$B_t^E = \sum_{L=L_{Min}}^{L_{Max}} s_{t,L} W_L N_{t,L}^E \quad (35)$$

where

$$N_{t,L}^E = G_{t,L} O_L^E N_{t-1,L}^E + E_L N_{t-1,L}^C \quad (36)$$

Catchability can be estimated analytically for each of the two periods 1985 – 1989, and 1990 – present, p , as:

$$q_p = \exp \left[\sum_{t=1}^{n_p} Ln \left(I_t / B_t^E \right) / n_p \right] \quad (37)$$

where n_p is the number of years in period p .

18.3.5.3 Likelihood Components

The model is fitted to catch rates and to the proportional catch at size data. For the catch rates, log-normal residual errors are used in a maximum likelihood framework. Predicted catch rates were generated from the exploitable biomass and the q estimates (there were two q estimates relating to the two time series of data (1985 – 1989, and 1990 – 2010); λ is set to 1.0):

$$\hat{I}_t = \frac{\hat{C}_t}{E_t} = q\hat{B}_t^\lambda e^\varepsilon \quad (38)$$

$$SSQ_p = \sum_{t=1}^{n_p} \left(\text{Ln}(I_t) - \text{Ln}(\hat{I}_t) \right)^2 \quad (39)$$

For each time series the negative log-likelihood for catch rates was estimated as:

$$LL_{CE} = \frac{n_p}{2} \left(\text{Ln}(2\pi) + 2\text{Ln}(\sigma_p) + 1 \right) \quad (40)$$

Where σ_p is defined as:

$$\sigma_p = \sqrt{\frac{SSQ_p}{n_p}} \quad (41)$$

The proportional length frequency data was included into the likelihood calculation uses multinomial likelihoods; again the negative log-likelihood was estimated as:

$$LL_{LF} = \sum_{t=1998}^{2008} -K_t \sum_{i=140}^{210} \frac{L_{t,i}}{\sum L_{t,i}} \text{Ln} \left(\frac{\hat{L}_{t,i}}{\sum \hat{L}_{t,i}} \right) \quad (42)$$

This summation is required across size classes 140mm – 210mm and for the years 1998 to 2010, with each yearly total being weighted by the square root of the number of observations ($K_t = N_t^{0.5}$).

To limit the variation in the recruitment residuals a penalty was added to the likelihood that increased as variation increased

$$P_R = \sum_{t=1}^n \text{Ln}(\varepsilon_R)^2 / 2\sigma_R^2 \quad (43)$$

A further penalty, equation (44), is applied to ensure that the initial harvest rate stays within possible bounds of 0 and 1 (**Figure 72**):

$$P_H = 100 * (\text{abs}(1 - \text{abs}(H_{init}) - 0.5) / 0.5)^{25} \quad (44)$$

The final total negative log-likelihood to be minimized, f , is designated as:

$$f = \text{WtCE} \times LL_{CE} + \text{WtLF} \times LL_{LF} + \text{WtRec} \times P_R + P_H \quad (45)$$

Table 27. Definitions and parameters for the three different periods of selectivity relating to the three periods of differing Legal Minimum Length (LML) on the west coast.

Start Year	Ls50	Ls95	LML
1985	127	132	127
1987	132	137	132
1990	140	145	140

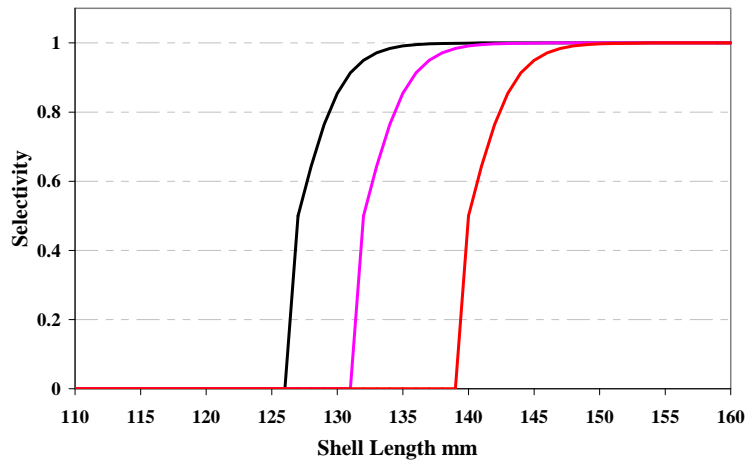


Figure 71. Selectivity during each of three periods in the fishery (Table 27). The left most curve relates to 1985 – 1986, the middle curve to 1987 – 1989, and the right hand curve to 1990 – 2008.

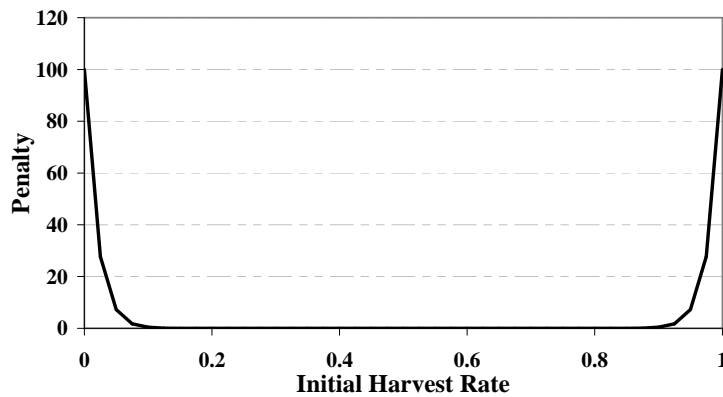


Figure 72. Penalty function applied to the initial harvest rate to ensure that it stays positive and does not extend beyond 1.0. See equation (44).

19 The Operating Model

19.1 The Operating Model Design

Not surprisingly the operating model used to describe the dynamics of a simulated abalone zone is made up of multiple populations, and its related fishery; the operating model has numerous components and appears relatively complex. Nevertheless, a simulated abalone zone has an hierarchical structure where a zone is made up of a number of spatial assessment units (SAUs), the statistical blocks of Tasmania, and these can contain a number of abalone populations of varying size, productivity, and other properties. The operating model is therefore structured to reflect this hierarchical form (**Figure 73**).

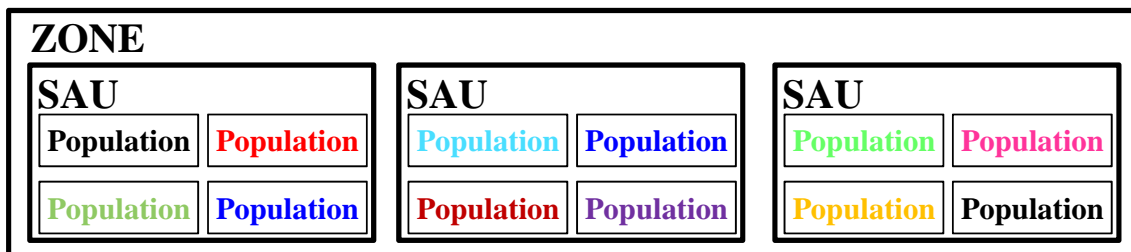


Figure 73. A diagrammatic representation of a simulated zone made up of a number of spatial assessment units (SAUs) each made up of a number of separate populations, each with its own properties (represented by the different toned colours). In an operational simulated zone there would likely be more SAUs and more populations in at least some of the SAUs and the populations are likely to be of very different sizes, each population with its own particular properties.

The fundamental unit within the operating model is therefore the population and this forms the basic building block within the simulation framework. To define a zone it is therefore necessary to define the number of populations across the complete zone, and then the number of SAUs these populations are to be grouped into and finally how they were to be grouped, that is, which population was to be a part of what SAU. For ease of calculation and consequent summary, each population, in the software, was designed to carry all the information required to define and characterize it (**Table 28**).

The spatial assessment unit structure was imposed on top of the population structure and the zone's properties, as well as its fishery, could be examined at a population level, at an SAU level, or at the whole zone level. The simulated data from the framework that could be used in any chosen assessment included the catch, the effort, the catch rate, and the spatial distribution of those statistics, and the size distribution of the catch (with pre-specified levels of uncertainty being included in such simulated data). These are the classical performance measures available in all Australian abalone fisheries; the size distribution of the catch is generally the least well known, although exceptions do exist where measuring boards are used as data loggers.

When comparing the performance of different management strategies it is also necessary to consider the state of the underlying stock. The data used to examine the state of the resource include the fishery data above but also the exploitable and spawning biomass (spawning and mature biomass are terms used interchangeably), the related harvest rates, the size distribution of the populations (rather than that of the catch), the re-

cruitment distributions, and the relative distribution of such things spatially. When not dealing with fisheries data there would be no uncertainty added to the available data.

Table 28. The component structure of each simulated population within the MSE simulation framework. Each population contains 37 objects, some of which are single numbers, others are vectors, and others are matrices. The variable names are those used in the software

Item	Variable	Values	Description
1	Max Δ L	1	Maximum growth increment from the Inverse Logistic
2	L50	1	Initial Size that generates 50% of maximum increment
3	L95	1	Initial Size that generates 5% of maximum increment
4	MaxSig	1	Maximum standard deviation around the growth increments
5	Me	1	Natural mortality of the emergent abalone
6	Mc	1	Natural mortality of the cryptic abalone
7	R0	1	Average unfished number of recruits
8	A0	1	Scales unfished spawning biomass with unfished recruits
9	B0	1	Unfished mature or spawning biomass
10	steeph	1	Steepness of the Beverton Holt Recruitment Curve
11	ExploitB	50	Exploitable biomass through the 50 years projection period
12	MatureB	50	Mature biomass through 50 year projection period
13	MatBCrypt	50	Mature biomass through 50 year projection period in crypsis
14	HarvestR	50	Annual Harvest rate in each of the 50 years
15	Catch	50	Annual catch from the population in each of 50 years
16	popdef	18	List of variables defining some of the population's properties
17	MSY	1	Dynamically estimated maximum sustainable yield.
18	MSYDepl	1	Stock depletion level where it generates the MSY
19	LML	1	Legal Minimum Length in the zone
20	bLML	1	Size at 50% maturity + 2 years growth: Biological LML or BML
21	popq	1	Population's catchability
22	SaM	1	Size at 50% maturity
23	cpue	50	Catch rate as kg/hr in each of 50 years
24	CatchN	105 x 50 (5250)	Numbers at size in catch from 2 - 210 for 50 years
25	deplExB	50	Depletion level of exploitable biomass in each of the 50 years.
26	deplSpB	50	Depletion level of spawning biomass in each of the 50 years.
27	Recruit	50	Absolute recruitment as numbers in each of 50 years
28	ExB0	1	Unfished exploitable biomass
29	G	105x105(11025)	Growth transition matrix
30	Maturity	105	Maturity ogive
31	WtL	105	Weight at Length relationship
32	Emergent	105x105(11025)	Square matrix describing emergence ogive
33	Select	105	Selectivity ogive
34	Nemerg	105 x 50 (5250)	Numbers at size in the emergent population in each year
35	Ncrypt	106 x 50 (5250)	Numbers at size in the cryptic population in each year
36	SelWt	105	Weight at Length x Selectivity ogive
37	MatWt	105	Weight at Length x Maturity ogive

19.2 The Operating Model Equations

19.2.1 Introduction

Any modelling framework can be described either using words or with more formal equations. Formal equations are far better for describing dynamic relationships between variables than words but in this report we will attempt to do both. In addition, some algorithms followed in the various analyses will be described using pseudo-code, which attempts to capture the sequence of processes required to run the simulations.

Each abalone fishery in Australia is split into regions or zones which are defined spatially (or by species, e.g. the greenlip fishery in Tasmania) and are allocated a specific total allowable catch (TAC). The MSE is designed to simulate a single zone and considers the dynamics of the fishery for the TAC across the various populations and SAUs within the zone.

Each simulated zone is made up of a pre-defined number of populations (defined in the global variable *numpop*), with a typical number of populations being 70 but it could be anything from 1 to 100s depending on the purpose of the simulation. The fundamental unit within the MSE framework is the population.

19.2.2 The Basic Dynamics

The dynamics of the simulations operate at an annual time scale and there is no distinction made between the sexes as they are deemed to grow in the same manner and are not distinguished by the divers. The size-structure that is used throughout uses 105 size classes of 2 mm from 2 – 210 mm, with the maximum size class acting as a plus group. This range covers the expected sizes to be found in Australia. The 2 mm (equivalent to sizes 1-3mm inclusive) was selected because the size at which the first shell pore becomes defined is generally somewhere between 1-2mm (Prince *et al*, 1988), and that is often deemed to be the start of the juvenile stage and to occur after two or three months (Cropp, 1989). The 210 mm plus group was selected because very few abalone, even in west coast Tasmania, grow larger than that. The 2mm size class was selected as a compromise between excessive computational load and fine detail in selectivity and growth.

While the sexes are combined there are, however, separate vectors of numbers-at-size for the cryptic and emergent components of the stock. The time step is annual with natural mortality being implemented in two halves with the remaining dynamics in between; details are given with the formal equations and the pseudo-code. All recruitment is into the cryptic component and any fishing mortality is imposed on the emergent component.

19.2.3 Structure of Each Simulated Population

Each population contains 37 objects (**Table 28**) some of which are single numbers, others are vectors, while others are matrices. In this way the complete character of each population can be captured or derived along with related fishery properties. Combining multiple populations into a single zone can lead to a data structure that can become quite large in terms of memory requirements and a reduced form can be saved instead which omits all the large matrices from the growth transition matrix down (omits rows 28 – 37 in **Table 28**).

Each population has a collection of variables concerning its growth, reproduction, natural mortality, biomasses, and its fishery. There are a few other items that are primarily there to facilitate and speed the calculation of the dynamics (for example the combination of weight at length with the selectivity and with maturity ogives and the use of a square matrix instead of a vector for the emergence ogive).

In order to apply the MSE framework it was necessary to include a spawning stock – recruitment relationship. There does not appear to be negative intra-cohort interactions, which excludes the Ricker stock-recruitment form so a Beverton – Holt relationship was developed (see equations).

19.3 Pseudo-code for the MSE Modelling

Define details of the Simulation

- the size structure used to describe dynamics
- the number of years for dynamics (max Nyrs = 50)
- the number of populations and the number of blocks
- the bounds and expectation for basic biology:
 - natural mortality,
 - growth,
 - recruitment,
 - size at maturity,
 - weight at length
 - emergence at size

Fishery details:

- Original TAC
- LML
- Selectivity at size
- Initial Exploitable biomass depletion level

Variability to be included in observed values of:

- Recruitment
- Distribution of biomass among SAUs or Populations
- Catch per unit effort
- Size distribution of the catch

Make a Simulated Zone

For each population select random values from within the general constraints for:

- Growth parameters
- Size at Maturity
- Recruitment relationship and variation
- Probability of zone-wide successful recruitment.
- Natural Mortality rates (emergent and cryptic).
- Weight at Length
- Migration ogive from cryptic to emergent

Define each population, its growth, and initiate its size structure

For the Years 2:Nyrs

Use function OneYear to generate the dynamics:

Recruitment, growth, fishing etc.

Storage of annual results into Zone list

Summarize Results

19.3.1 Model Variables

- a, b the weight at length parameters,
 α, β are the maturity at length logistic parameters, α/β depicts the length at 50% maturity and $2.Ln(3)/\beta$ is the inter-quartile distance,
A the complement of an annual harvest rate (via a selectivity curve, s) applied to the emergent animals, $(\mathbf{I}-\mathbf{sH})$
C_s a square zero matrix ($n \times n$) with the survivorships, $e^{-(M/2)}$, by size class down the diagonal elements for cryptic abalone, this involves the survivorship from half the natural mortality (which need not be the same as for emergent abalone,
E a square zero matrix ($n \times n$) with the proportion emergent by size class, Eq (52), arranged along the diagonal elements,
G a square growth transition matrix ($n \times n$), the same for both sexes,
H Annual harvest rate,
I the unit matrix,
 I_t the standardized catch rate in year t .
 L_{E50} logistic parameter for the emergence curve, depicts the length at which 50% of cryptic animals become emergent,
 L_{E95} logistic parameter for the emergence curve, depicts the length at which 95% of cryptic animals become emergent,
 $\bar{L}_{i,j}$ the expected mean length of animals starting in size class j ,
 L_{m50} logistic parameter for the growth curve, depicts the length at which the growth increment is 50% of the maximum,
 L_{m95} logistic parameter for the growth curve, depicts the length at which the growth increment is 5% of the maximum,
 L_{min} the minimum size class considered,
 L_{max} the maximum size class considered,
LML legal minimum length
 L_{s50} logistic parameter for the selectivity curve, depicts the length at which 50% selection occurs,
 L_{s95} logistic parameter for the selectivity curve, depicts the length at which 95% selection occurs,
LW the class width in mm,
 M natural mortality, which can be different in crypsis and emergent populations,
Max ΔL the maximum growth increment for the inverse logistic curve describing abalone growth, The point at which variation is 5% of the maximum is set at 210mm for the west coast and the 50% point is set at L_{m95} ,
Max σ_L the maximum standard deviation describing the variation around the mean expected growth increment,
 m_L maturity at length L ,
 \mathbf{N}_t^C a vector of numbers-at-size in year t for cryptic abalone, with n size classes,
 \mathbf{N}^{C*} the equilibrium initial population size structure for cryptic animals,

- \mathbf{N}_t^E a vector of numbers-at-size in year t for emergent abalone, with n size classes,
- \mathbf{N}^{E*} the equilibrium initial population size structure for emergent animals,
- origTAC The TAC at the start of a simulation; allows for repeating the analysis assuming that the control rule used alters the active TAC during each run,
- O_s a square zero matrix ($n \times n$) with the survivorships, $e^{-(M/2)}$, by size class down the diagonal elements for emergent abalone, this involves the survivorship from half the natural mortality (which need not be the same as for cryptic abalone,
- q the catchability,
- \mathbf{R} a vector (n) of recruitment numbers (generally zero except for the smallest size classes),
- \mathbf{S} a square zero matrix ($n \times n$) with the survivorships, $e^{-(M/2)}$, by size class down the diagonal elements for emergent abalone, this involves the survivorship from half the natural mortality,
- s_L selectivity of length class l ,
- σ^j the expected standard deviation for length class j ,
- σ_R^2 the variance of the recruitment residuals,
- TAC total allowable catch (see origTAC)
- W_L the weight in grammes of abalone of length L ,
- $WiCE$ the weight given to the catch effort contribution to the negative log-likelihood,
- $WiLF$ the weight given to the proportion length frequency data to the negative log-likelihood,
- $WiRec$ the weight given to the penalty on recruitment variation,

19.3.2 Model Initiation

The mortality schedules differ between the cryptic and emergent population components because a constant initial fishing mortality is applied to the emergent population and exactly how the fishing mortality is implemented in the model needs to be reflected in the equilibrium equations. The model structure adopted has half of natural mortality occurring followed by growth and fishing mortality, followed by the remaining natural mortality. If natural mortality is implemented as half natural mortality, that is

$C_s = e^{-M/2}$ for cryptic and $O_s = e^{-(M/2)}$ for emergent, twice in the year, with other dynamics between the natural mortality events then the dynamics, first for the emergent numbers at size and then for the cryptic numbers at size can be represented as:

$$N_{t+1}^E = O_s \left[GO_s (N_t^E + EN_t^C) \right] \quad (46)$$

and
$$N_{t+1}^C = C_s \left[GC_s (N_t^C - EN_t^C) \right] + R \quad (47)$$

at equilibrium
$$N^{C*} = \left(I - [C_s GC_s (I - E)] \right)^{-1} R \quad (48)$$

Consequently, for emergent abalone:

$$N^{E*} = (I - O_s GO_s)^{-1} O_s GO_s EN^{C*} \quad (49)$$

If there is an initial estimated fishing mortality rate, this can be defined as the complement of an annual harvest rate and is distributed down the diagonal of an otherwise zero square matrix A :

$$A_L = (1 - s_{L,t} H_t) \quad (50)$$

where A_L is the survivorship of length class L , $s_{L,t}$ is the selectivity of length class L in year t (which relates to the LML), and H_t is the fully selected harvest rate in year t . With an initial fishing mortality rate there would be no change to the equilibrium for the cryptic component, Eq (48), but the equilibrium numbers for the emergent population would become:

$$N^{E*} = \left((I - O_s A G O_s)^{-1} (O_s A G O_s EN^{C*}) \right) \quad (51)$$

19.3.3 Biology and Stock Related Statistics

Transfer from crypsis into emergence is described using a standard logistic equation (Haddon, 2011):

$$E_L = \frac{1}{1 + e^{-Ln(19)(L-L_E 50)/(L_E 95-L_E 50)}} \quad (52)$$

Where E is the proportion of size class L that are emergent, and $L50$ and $L95$ are the usual logistic parameters defining the lengths at which 50% and 95% are emergent.

The growth from size-class to size-class is described by the elements of a growth transition matrix defined by:

$$\begin{aligned}
G_{i,j} &= \int_{-\infty}^{L_i + \frac{LW}{2}} \frac{1}{\sqrt{2\pi}\sigma^j} e^{-\left[\frac{(L_i - \bar{L}_{i,j})^2}{2(\sigma^j)^2}\right]} dL & L_i = L_{Min} \\
G_{i,j} &= \int_{L_i - \frac{LW}{2}}^{L_i + \frac{LW}{2}} \frac{1}{\sqrt{2\pi}\sigma^j} e^{-\left[\frac{(L_i - \bar{L}_{i,j})^2}{2(\sigma^j)^2}\right]} dL & L_{Min} < L_i \leq L_{Max}
\end{aligned} \tag{53}$$

to ensure that all columns sum to 1.0 and to make L_{Max} a plus group the final row of the matrix is modified for each column j as:

$$G_{L_{Max},j} = G_{L_{Max},j} + \left(1 - \sum_{i=L_1}^{L_{Max}} G_{i,j}\right) \tag{54}$$

An alternative approach would be to include a very large number as the upper bound of the last size class. The expected mean size for each size class j is defined using an inverse logistic growth curve that has been found to describe blacklip abalone growth well (Haddon *et al.* 2008; Helidoniotis *et al.*, 2011):

$$\bar{L}_{i,j} = L_j + \frac{Max\Delta L}{1 + e^{\frac{Ln(19)(L_j - L_m 50)}{(L_m 95 - L_m 50)}}} + \mathcal{E}_{L_j} \tag{55}$$

Variation around the mean expected growth increment is assumed to be normally distributed with a standard deviation that varies with the growth increment (Haddon *et al.* 2008):

$$\sigma_{L_j} = \frac{Max\sigma_L}{1 + e^{\frac{Ln(19)(L_j - L_m 95)}{(210 - L_m 95)}}} \tag{56}$$

The weight at size, W_L , relationship

$$W_L = aL^b \tag{57}$$

Maturity at size, m_L ,

$$m_L = \frac{e^{(\alpha + \beta L)}}{1 + e^{(\alpha + \beta L)}} \tag{58}$$

and selectivity for length L in year t is defined as:

$$s_{L,t} = \frac{1}{1 + e^{-Ln(19)(L - L_s 50)/(L_s 95 - L_s 50)}} \tag{59}$$

Mature or spawning biomass needs to include contributions from both the emergent and cryptic components of the population (thus numbers at size by maturity at size and weight at size):

$$B_t^S = \sum_{L=L_{min}}^{L_{max}} (N_L^E m_L W_L) + \sum_{L=L_{min}}^{L_{max}} (N_L^C m_L W_L) \tag{60}$$

Exploitable biomass is estimated after half of natural mortality, growth, and emergence (divisible by 1,000,000 to estimate as tonnes) and before any fishing mortality occurs in any single year. Only emergent biomass is considered as no fishing mortality is imposed on the cryptic component:

$$B_t^E = \sum_{L=L_{Min}}^{L_{Max}} s_{t,L} W_L N_{t,L}^E \quad (61)$$

where

$$N_{t,L}^E = G_{t,L} O_L^E N_{t-1,L}^E + E_L N_{t-1,L}^C \quad (62)$$

Catchability in a stock assessment model can be estimated analytically as:

$$q = \exp \left[\sum_{t=1}^n \text{Ln} (I_t / B_t^E) / n \right] \quad (63)$$

where n is the number of years across which the observed catch rates and predicted exploitable biomass are considered. In the simulation model a maximum catch rate, CE_{Max} was used to scale the unfished exploitable biomass to generate a catchability value for each population. The maximum catch rates were randomly selected from a pre-specified distribution, and then the following equation used:

$$q_p = (CE_{Max,p} / B_{0,p}^E) \quad (64)$$

Where the index is for each population p and B_0^E is the unfished exploitable biomass.

19.3.4 Model Dynamics

Once each population is initiated its dynamics can be projected forwards a year at a time depending on how much catch is expected to be taken or how much effort expected to be focussed into each population. The initiation sets up the equilibrium numbers for the initial conditions established for each population. Then given a specific harvest rate for the each population they can be projected forward in yearly steps. This projection is based around what is expected to occur to the numbers of animals in crypsis and then the number of animals emergent. As before, if the fishing mortality rate over a year is defined as the complement of an annual harvest rate and is distributed down the diagonal of an otherwise zero matrix \mathbf{A} :

$$A_L = (1 - s_{L,t} H_t) \quad (65)$$

where A_L is the survivorship of length class L , $s_{L,t}$ is the selectivity of length class L in year t , and H_t is the fully selected harvest rate in year t . And if natural mortality is implemented as half natural mortality, that is $\mathbf{C}_S = e^{-M/2}$ for cryptic and $\mathbf{O}_S = e^{-(M/2)}$ for emergent, twice in the year, with other dynamics between the natural mortality events then the dynamics can be represented as:

$$\mathbf{N}_{t+1}^E = \mathbf{O}_S \left[\mathbf{G} \mathbf{A}_t \mathbf{O}_S (\mathbf{N}_t^E + \mathbf{E} \mathbf{N}_t^C) \right] \quad (66)$$

and

$$\mathbf{N}_{t+1}^C = \mathbf{C}_S \left[\mathbf{G} \mathbf{C}_S (\mathbf{N}_t^C - \mathbf{E} \mathbf{N}_t^C) \right] + \mathbf{R} \quad (67)$$

19.3.5 Stock Recruitment Relationship

Punt (2003) includes a Beverton & Holt stock recruitment relationship in a size-structured model designed to work with southern rock lobster, and he writes of the two parameters (alpha and beta) being re-parameterized in terms of steepness, h . This relat-

ed to work by Francis (1992) who re-parameterized the Beverton-Holt curve into terms of steepness for age-structured models. This re-parameterization is general across both age-based and size-based models. Recruitment is added to the contents of the first size class (2mm), all other size classes being set to zero.

The size-based equivalent to Francis' (1992) re-parameterization requires the assumption that an unfished population under constant recruitment will achieve a constant size distribution (and presumably a constant age distribution but this remains unknown for abalone). From the constant size distribution it is possible to calculate B_0 , the total mature biomass found in the unfished population, which is the equivalent of eq (60) except uses the unfished equilibrium numbers at size eqs (49) and (51). Thus:

$$B_0 = \sum (N_L^{E*} m_L W_L) + (N_L^{C*} m_L W_L) \quad (68)$$

which would be the spawning biomass at the start of each year. This equation translates between the equilibrium size distribution produced by the virgin average recruitment level, R_0 , into the unfished mature biomass B_0 . Francis (1992) used this relation to develop a direct scaling parameter A_0 , which was the mass of mature biomass produced at equilibrium from a constant single recruit. By combining this with the virgin recruitment level R_0 a direct estimate of the unfished mature biomass could be produced:

$$B_0 = R_0 A_0 \quad (69)$$

The virgin mature biomass per recruit generated by a constant recruitment level of one (A_0) can be obtained using eqs (66) to (68) with recruitment in eq (67) set to 1.0. With an estimate of A_0 the recruitment levels from plausible levels of B_0 can be obtained from eq (69). Francis' (1992) re-parameterization consisted of re-parameterizing the Beverton-Holt parameters thus:

$$\alpha = \frac{B_0(1-h)}{4hR_0} \quad \text{and} \quad \beta = \frac{5h-1}{4hR_0} \quad (70)$$

Punt (2003) used the classic Beverton and Holt equation that used these estimates of α and β , however, the R_0 value can be used directly as in Haltuch et al. (2008):

$$N_{t,0} = \frac{4hR_0 B_t^M}{(1-h)B_0 + (5h-1)B_t^M} e^{\varepsilon_t - \sigma_R^2/2} \quad (71)$$

The $\varepsilon - \sigma_R^2/2$ term is there to allow for bias in the log-normal relationship so that the simulated recruitments relate to the median of the distribution rather than the mode.

In the simulations the stock-recruitment relationship for each population can thus be defined in terms of steepness and by simulating either an unfished virgin recruitment level or an unfished biomass level, with the required values being sampled from predetermined distributions.

19.4 Conditioning the Operating Model

19.4.1 Introduction

The abalone section of the former Tasmanian Fisheries and Aquaculture Institute, and now the Institute for Marine and Antarctic Studies, has, through the last 20 years, developed a database of observations on different abalone populations around Tasmania.

It is well understood that, in abalone, there is a great deal of spatial variation in the biological attributes relating to growth, morphology, and maturity. Many studies have confirmed this (Worthington *et al.*, 1998; *etc.*), in Tasmania while there are numerous samples the full extent of variation is still to be documented (although variation in size at maturity is an exception (Tarbath *et al.*, 2001); although see **Error! Reference source not found.** Helidoniotis and Haddon, (2013). Information relating to the morphometrics will also be considered, specifically the weight at length relationship. In addition, the growth of abalone is highly variable around Tasmania and is an important component of size-based models. Finally, maturity is obviously important in the estimation of the spawning biomass in any population and so how it varies is also important. While emphasis will be placed on characterizing the variation and functional form of these biological properties attention will also be paid to how best to simulate the range of variation so as best to condition the operating model in the Management Strategy Evaluation framework.

19.4.2 Length to Weight Relationship

In a size-based model to convert a size distribution to a measure of mass requires a length to whole weight relationship. This typically forms a power relationship of the form:

$$Wt_L = aL^b e^\varepsilon \quad (72)$$

Where the two parameters a and b relate length to weight; the expected residual error structure is log-normal. This means that this relationship can be fitted to log-transformed length and weight data by using simple linear regression (Figure 74). There were 122 sites around Tasmania where data had been collected on both shell morphology and body weight.

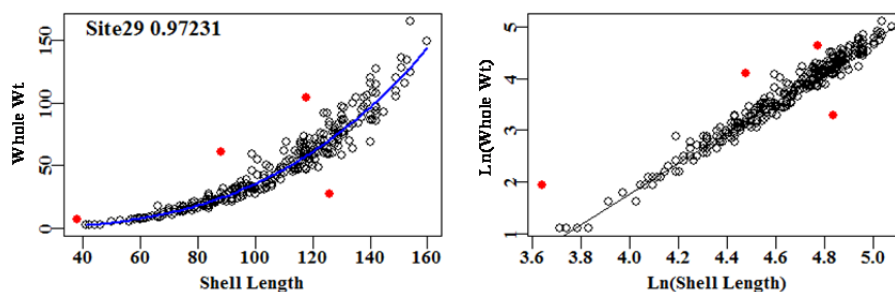


Figure 74. A typical weight at length relationship using data from sites around Tasmania (site 29 is on the Tasman Peninsula, Long = 147.38, Lat = -43.11). The 0.972 is the R^2 for the regression. The red points were outliers detected as greater than three times the expected standard deviation of the residuals for each length. For site 29 the $a = 2.7937e-05$ and $b = 3.3294$.

When the parameters for all 122 sites are plotted against each other (Figure 75) it is clear that they too are connected in a power relationship akin to Equ (72).

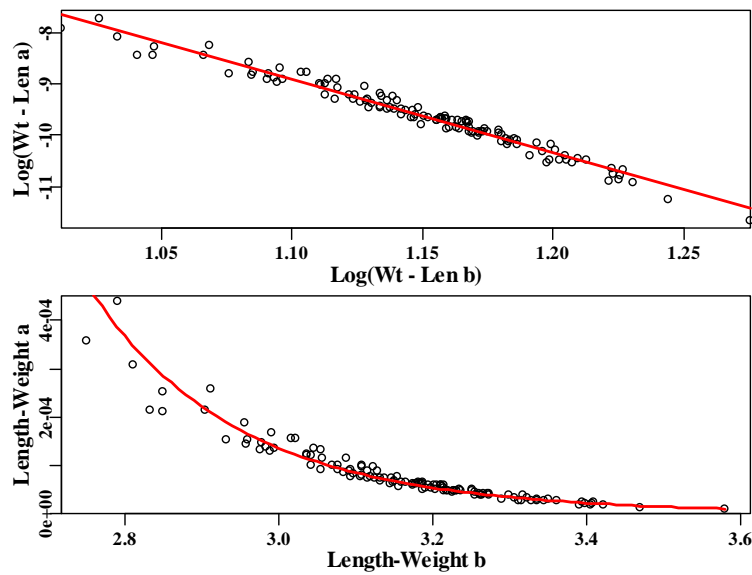


Figure 75. Relationship between the a and b parameters of the Weight at Length relationships for 122 sites around Tasmania. The relationship is $a = 962.8098 \times b^{-14.35264}$, with $R^2 = 0.968$.

It is well known that some areas yield higher recovery rates in terms of meat weight relative to total weight than other areas. A reflection of this can be seen when the trajectories relating weight to length for the 122 sites are considered (Figure 76), with some animals at 160 mm having an average weight of only about 375 g while others of the same size can weigh up to 750 g. There would be variation around each curve.

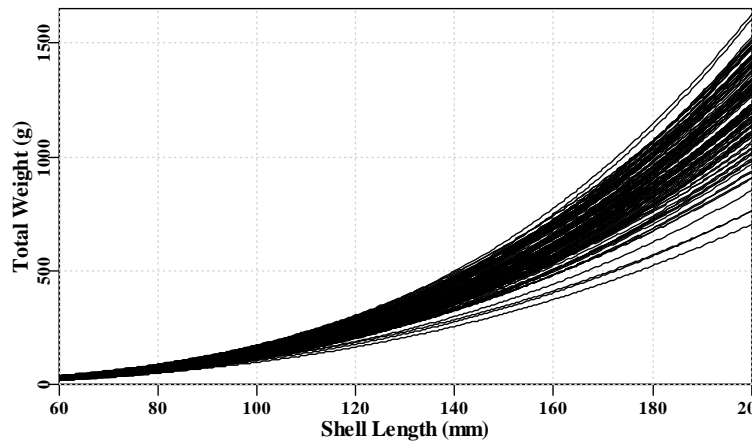


Figure 76. The trajectories of all 122 weight at Length relationships as observed around Tasmania. The weight to length relationship can clearly influence the total weight of animals. In the operating model, to simulate the variation in the weight at length relationship first the b parameter was randomly picked from a normal distribution with mean = 3.162 and standard deviation = 0.1485 (Figure 77).

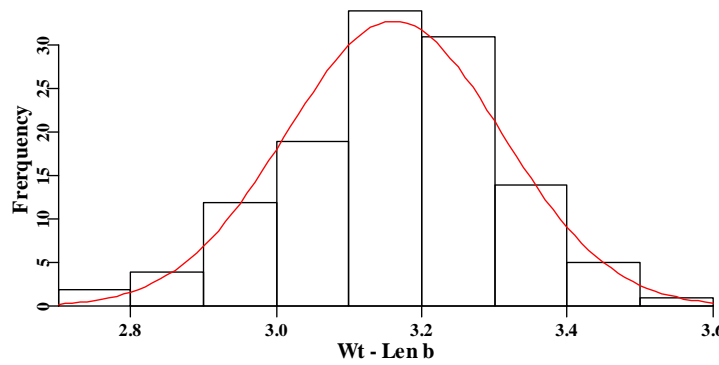


Figure 77. The observed distribution of the b parameter from the weight at length power relationship of 122 sites from around Tasmania. The mean = 3.161963 and the standard deviation = 0.1484613 was used to generate the red line.

To simulate a weight at length relationship for each population in a simulated zone a random value is selected from the distribution of b parameters (**Figure 77**) and the equation relating the a parameter to the b value is used to further define the required parameter needed for the weight at length relationship.

19.4.3 The Simulation of Maturity Ogives

The proportion of any given size class that is expected to be mature can be described using a logistic equation:

$$\hat{m}_L = \frac{e^{(a+bL)}}{1 + e^{(a+bL)}} \quad (73)$$

Where the two parameters are a and b , such that the shell size where 50% of individuals are expected to be mature, the size-at-maturity is defined as:

$$SM_{50} = \frac{-a}{b} \quad (74)$$

And the interquartile distance, which provides an index of the rapidity with which maturity progresses as size increases is defined as:

$$IQ = \frac{2Ln(3)}{b} \quad (75)$$

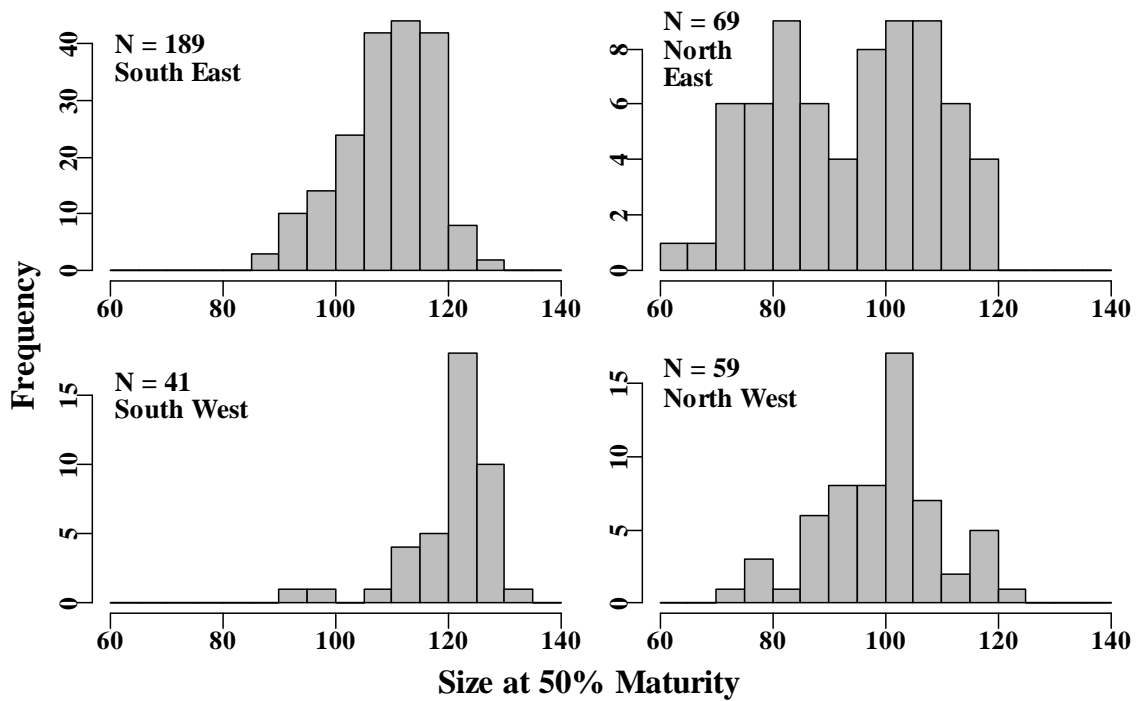


Figure 78. The distribution of size at maturity, defined as the size at which 50% of a population is mature, for the east coast (Longitude > 146.5°) and west coast, and the north and south (split at -42°), with the number of observations for each. 358 samples in total.

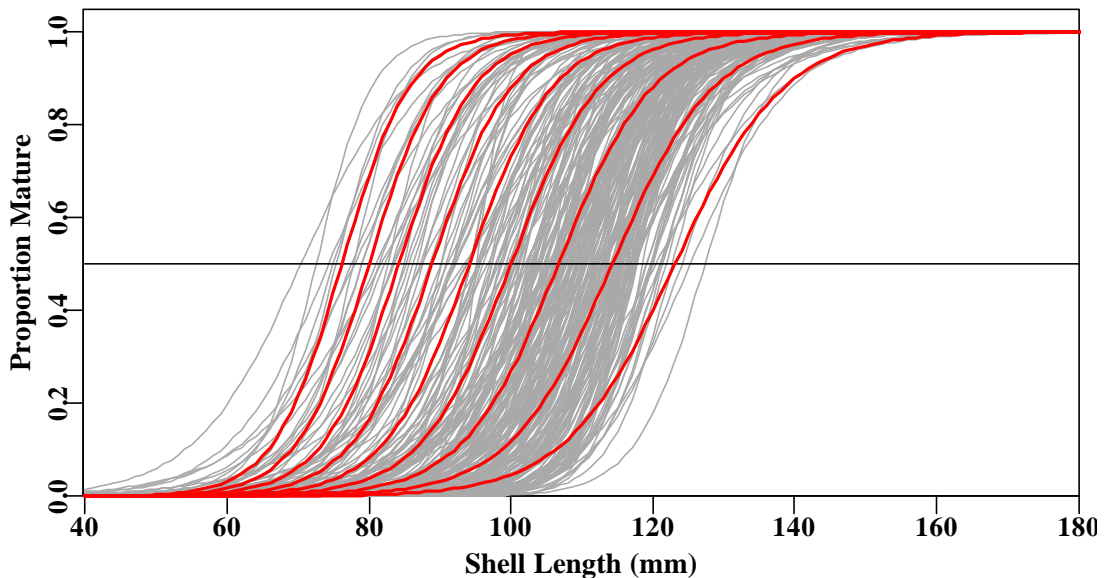


Figure 79. The grey lines represent the array of size at maturity ogives for the east coast, with 50% maturity points ranging from about 65 – 127 mm. The relative density of lines reflects the relative frequency of occurrence of each set of values, although the intensity of sampling has not always been evenly spread along the coast. The red lines are 9 different maturity curves defined using Equ (73), with an a parameter of -16 and a b parameter varying from 0.13 (rightmost curve) to 0.21 (leftmost curve); in the simulations values of b were selected in proportion to the relative frequency seen in samples and then additional noise added so that the full range of variation was expressed in the simulations. See **Table 30** for the properties of each curve. The horizontal black line is at 50%.

Table 29. Growth parameters and other properties of the 27 sites around Tasmania examined. EW was split at 147.5. The sites are sorted by EW and then by Latitude (Lat). MaxObsL is the largest animal in the sample fitted.

Site	MaxDL	L50	L95	MaxSig	Nobs	MaxObsL	Lat	Long	Block	EW
315	19.63560	86.55890	120.45518	4.03424	207	135.9	-39.69	147.88	37	E
316	24.58234	97.11246	144.40851	4.51195	232	147.9	-40.73	148.12	31	E
588	19.73968	125.92142	164.79502	3.53868	118	181.2	-40.92	148.32	31	E
59	27.28685	113.95734	161.37465	4.92848	499	159.7	-41.57	148.32	29	E
300	32.32707	123.21436	160.66523	5.48833	114	156	-41.74	148.30	29	E
297	25.98925	114.67848	153.05017	5.25666	271	151.7	-42.20	148.35	27	E
159	19.51636	131.39260	157.89083	6.57909	210	172.9	-42.58	148.05	24	E
337	17.45597	107.59627	135.32075	5.03983	144	146	-42.87	147.94	23	E
764	24.19526	124.85150	170.05236	3.55467	226	160.6	-43.14	147.68	20	E
663	35.49251	87.44243	132.54669	4.87369	114	162	-43.04	147.48	17	W
482	26.87347	117.81048	153.97894	4.59842	135	155.8	-43.11	147.40	16	W
461	27.91589	122.21163	170.56414	3.89385	163	157.5	-43.11	147.38	16	W
702	26.94043	121.96601	171.65201	4.90053	257	173	-43.14	147.39	16	W
813	12.59615	111.46131	127.18670	4.02551	167	159	-43.51	146.98	14	W
478	22.33285	108.94727	145.78452	4.00345	347	149	-43.54	146.99	13	W
815	13.47087	117.85039	144.77014	3.62134	62	154.3	-43.53	146.96	13	W
480	31.06347	95.36221	136.32528	5.08516	151	145	-43.56	146.89	13	W
459	15.58214	128.41674	154.09631	3.42573	132	167.1	-43.48	146.02	12	W
470	21.73676	127.74274	163.37726	4.42795	55	179	-43.22	145.78	11	W
460	20.25673	129.45886	162.35022	5.15331	90	173.9	-43.07	145.66	11	W
458	19.19153	131.60463	164.04494	3.89195	118	171	-42.96	145.49	10	W
272	22.85316	122.96847	160.57304	4.23480	203	182	-42.61	145.26	10	W
819	19.67947	106.48766	139.88758	3.92818	97	158.9	-41.76	145.00	7	W
170	12.07867	115.96866	145.49746	3.34417	92	150.7	-41.17	144.67	6	W
313	16.03189	95.26875	126.10463	4.28612	389	145	-40.50	144.70	49	W
662	6.84730	81.81876	96.65249	3.73206	112	112.2	-40.86	145.51	46	W
314	22.37116	110.21989	147.57671	5.10382	434	161	-39.93	143.83	3	W

Table 30. The properties of maturity ogives as the b parameter is altered from 0.1 to 0.2; both the size at maturity and the inter-quartile distance changes. See Figure 79.

a	b	SM50	IQ
-16	0.12	133.333	18.310
-16	0.13	123.077	16.902
-16	0.14	114.286	15.694
-16	0.15	106.667	14.648
-16	0.16	100.000	13.733
-16	0.17	94.118	12.925
-16	0.18	88.889	12.207
-16	0.19	84.211	11.564
-16	0.20	80.000	10.986
-16	0.21	76.190	10.463

19.4.4 The Simulation of Growth Parameters

Growth is one of the biological characteristics of abalone that varies markedly between populations. When simulating a coastal fishery some method is required for allocating sets of growth parameters to each population. The optimum growth model for the description of growth in blacklip abalone in Tasmania has been found to be the Inverse Logistic (Haddon *et al.*, 2008; Helidoniotis *et al.*, 2011). The three main parameters of the Inverse Logistic curve, used to describe growth increments in blacklip abalone, are all correlated to some extent and this fact can be used in the simulation of plausible growth curve parameters. There are 27 sites at which annual tagging data have been collected and the growth curve estimates from these sites were used as the training data to populate the relationships between parameters that were used to generate plausible sets of parameters (**Figure 82**; **Table 29**).

By first simulating typical values for the L50 parameters the rest of the required growth parameters can be produced. Two specific relationships were used when generating arrays of growth parameters. The first was the linear relationship between the L95 and the L50 parameters (**Figure 80**).

$$L95 = 29.8537 + 1.0487 \times L50 \quad (76)$$

By first generating a random collection of L50 values from a typical distribution of such values, these can be used to generate related L95 estimates by using Eq. (76) and adding normal random error to each of the expected mean L95 values. Starting with the original data this process generates pairs of values that loosely resemble the distribution of the training values (**Figure 80**).

The second relationship used was the linear relationship between all three parameters;; Eq (77). The addition of the MaxDL data to this relationship significantly improved the fit of the statistical model (**Figure 81**).

$$L95 = 7.866 + 1.031L50 + 1.107MaxDL \quad (77)$$

The distribution of residuals above and below the plane of the bivariate relationship, Equ (77), exhibited no obvious pattern (Figure 81); though the smallest MaxDL also has the smallest values for L50 and L95 (site 662). Of all the sites whose growth has been measured/estimated, site 662 has the lowest productivity. Site 662 is in the Tasmanian Bass Strait zone which as a LML of 114 mm and since 2002 only has a reported catch

varying from 0 – 5 tonnes (the reported catch was zero from 1990 – 2001). In the simulation of growth parameters for a given coast an explicit decision needs to be made whether to include this extreme.

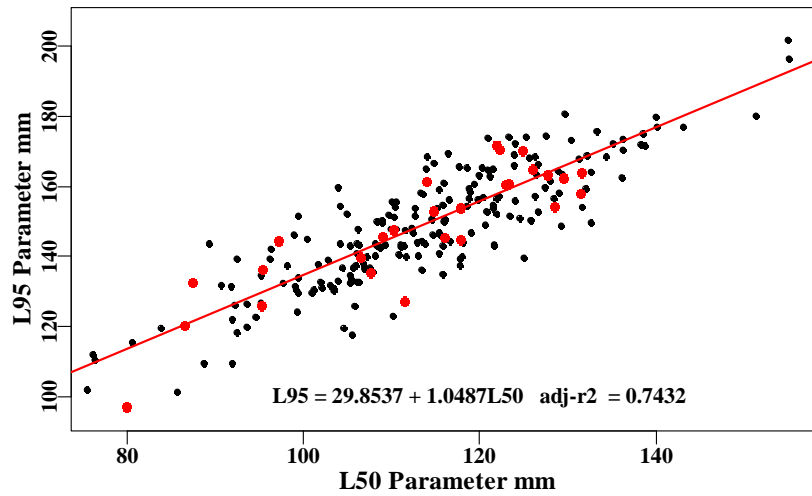


Figure 80. Relationship between the L95 and L50 parameters of the Inverse Logistic growth model as estimated from 27 populations around Tasmania. This relationship was used when generating parameters to describe the growth of the simulated abalone populations. The red dots are the original data while the black dots are produced by randomly selecting 200 L50 values using the linear relationship to generate the mean expected with a mean of 113 and 148 mm for L50 and L95 respectively. The variance covariance matrix was $c(213.9722, 224.3954, 224.3954, 312.4736)$.

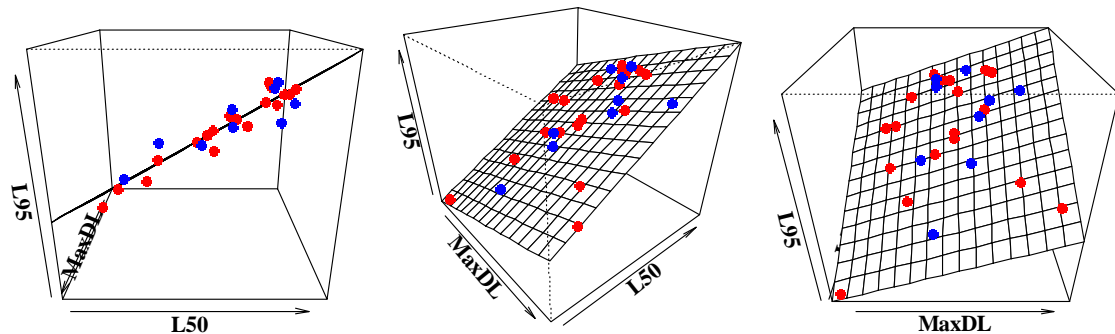


Figure 81. Plot of the 27 sets of growth parameters available. The relationship was fitted to the data as $L95 \sim L50 + MaxDL$; the estimates were $L95 = 7.86604 + 1.03107L50 + 1.10695MaxDL$, which leads to an $adj-r^2 = 0.9118$, $P = 8.522e-14$. See Eq. (77). Blue dots are from sites to the east of longitude 147° while red dots are those sites to the west.

Table 31. Statistical outcome of fitting the linear surface between $L95 \sim L50 + MaxDL$; there were highly significant parameter estimates for the variable although not the intercept.

Parameter	Estimate	StErr	t-value	P
Intercept	7.86604	8.62819	0.912	0.371
L50	1.03107	0.07044	14.637	1.85e-13
MaxDL	1.10695	0.15854	6.982	3.21e-07

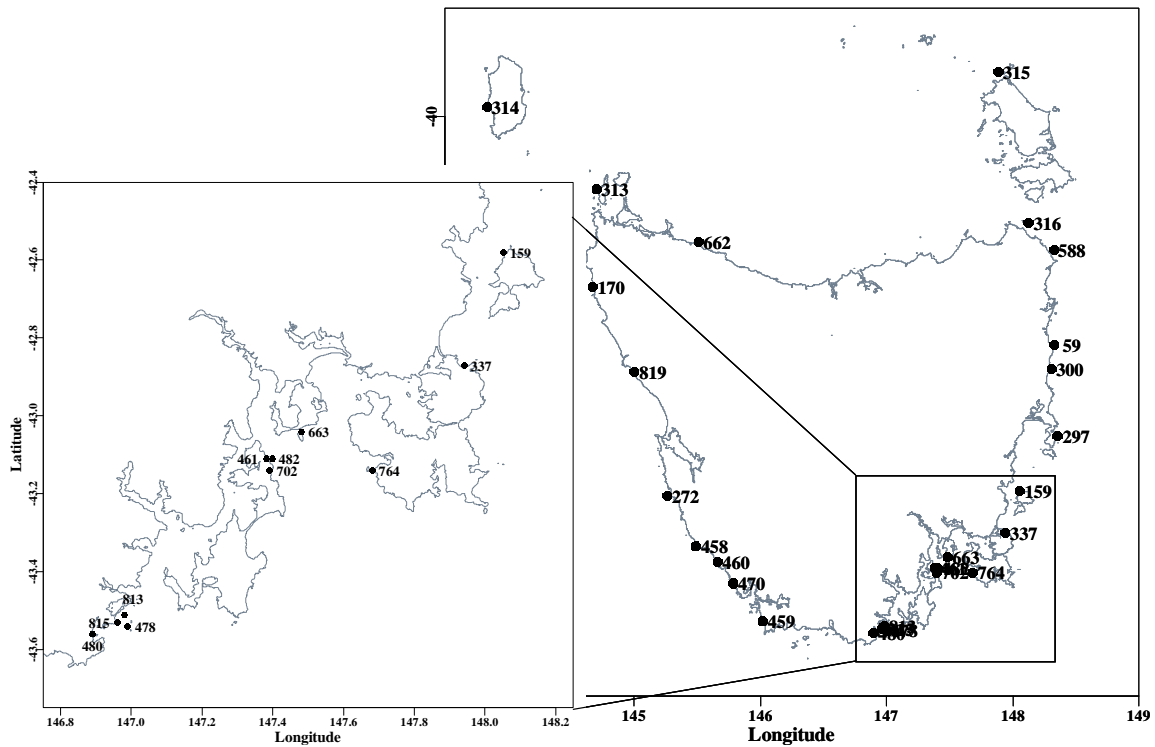


Figure 82. Map of the location of the 27 tagging sites for which Inverse Logistic growth parameters are available.

19.4.5 The Size Transition Matrix

The main driver of the dynamics of this modelling strategy is the size transition matrix, which describes the expected growth of individuals from one set of size classes into the next set of size classes. Given estimates of the growth curve parameters, perhaps from a tagging study, it is simply to generate the required matrix. To allow for more detail it would be better to have an underlying seasonal model but sufficient data to generate a seasonal growth model is only available at three sites around Tasmania. However, we can use that data to indicate that about two thirds of annual growth occurs in the first half of the year and the remainder in the rest of the year. This seasonality can be estimated by first estimating the transition matrix using the given growth parameters except the MaxDL parameter is set at 2/3 of the original yearly value. For the yearly growth description:

$$\mathbf{G} \mid \text{MaxDL}, L50, L95, \text{MaxSig} \quad (78)$$

These same parameters are used to describe up to 2/3 of the yearly growth occurring in the first size months:

$$\mathbf{G2} \mid 0.666\text{MaxDL}, L50, L95, \text{MaxSig} \quad (79)$$

The transition matrix for the remaining six months can be obtained by:

$$\mathbf{G3} = \mathbf{G.G2}^{-1} \quad (80)$$

There is a limit to what proportion of the yearly growth can be focused in the first six months, beyond which it is not possible to find a valid inverse matrix. Empirically, the more rapid period of growth in examples where seasonal growth descriptions can be

fitted (Haddon *et al.*, 2008) accounts for about 2/3rd of the yearly growth, in terms of the MaxDL value. Examining this option for the dynamics in the future could be possible.

19.5 Fishing a Simulated Abalone Zone

The essence of the management strategy evaluation is that a simulated abalone zone can be manipulated into different initial conditions of depletion and then fished for a given number of years under different initial conditions and with different management arrangements in place. Each combination of zone parameters, initial depletion status, monitoring data, performance measures, and control rules used, initial TAC, and LML constitutes a single scenario. The simulations are repeated many times for each scenario chosen and in this way comparisons of the distributions of outcomes of interest can be compared. A number of variables need to be set including the inherent variability of the distribution of catches among separate areas and the variation expected in observed catch rates and how catch rates are expected to vary relative to exploitable biomass. It is possible, for example, to vary such things as the linearity or otherwise of the relationship between exploitable biomass and catch rates.

In the Tasmanian abalone fishery a linear relationship between catch rates and catches, and hence with effort, is often observed. Because of this catch rates are assumed to have some influence over the distribution of catches among areas. Observed catch rates (CE) would naturally be expected to be variable and so are modelled as:

$$CE = q_a B_{t,a}^\lambda e^{N(0,\sigma_q)} \quad (81)$$

where q_a is the catchability coefficient exhibited in area a , $B_{t,a}^\lambda$ is the exploitable biomass in area a at time t , with a non-linearity coefficient of λ , $e^{N(0,\sigma_q)}$ is a log-normal random deviate, and σ_q is the standard deviation of the catchability coefficient q ; if λ is set equal to one then the relationship between catch rates and exploitable biomass is linear; this is the standard assumption in this work, although alternatives have been investigated in the study of the surplus production modelling this is not reported here. Given a TAC the catches will be distributed among the different areas in any given year. Catch rates are often assumed to provide an index of relative abundance and thus, previous catch rates may be considered able to serve as a guide to where to fish in subsequent years. This is a reasonable assumption if the fishery regularly leaves behind a significant proportion of the legal sized animals. However, catch rates in one year do not give any indication of the availability of undersized animals that are expected to grow into the fishery. In fisheries that are being fully exploited the advent of new recruits will be an important component of each year's fishery. Fortunately, the abalone fishery depends on divers literally handling their catch and this automatically provides them an opportunity to identify visually those areas that would be expected to be productive in the next year and also those areas that would be expected to become less productive. Their own observations would be made with some error and discussions among divers would also rarely be precise. Such diver expectations provide an indication of exploitable biomass and this can be used in an algorithm for distributing catches by area:

$$C_{t,a} = TAC \frac{B_{t,a} e^{N(0,\sigma_b)}}{\sum_{a=1}^n B_{t,a} e^{N(0,\sigma_b)}} \quad (82)$$

where $C_{t,a}$ is the expected catch in area a in year t , TAC is the total allowable catch, and σ_b is the standard deviation of the catchability interpreted as the diver's observations on available biomass in the n areas assessed within the year in question.

This approach reflects the system adopted in Dichmont *et al* (1999) and by Dichmont and Brown (2010) for distributing a TAC among areas. Their approach was related directly to catch rates (despite their equation implying a catchability coefficient of 1.0), however, the exploitable biomass is directly related to catch rates and so, especially with the random noise added to the biomass values this can adequately drive the distribution of catches.

As these proxies are for the diver perception of relative abundance they automatically include their knowledge of catches and catch rates from previous years. This approach can be used directly on the separate populations or, more in line with how the zones are managed, to collections of populations, known as statistical reporting blocks in Tasmania or Spatial Assessment Units (SAUs) more generally. It would be expected that as σ_b increased the ability of divers to appropriately distribute catches between areas would decline which would, in turn, be expected to lead to poor outcomes for the fishery in terms of depletion levels within blocks.

19.6 Productivity of a Simulated Zone

19.6.1 Introduction

Just as the dynamics of each population can be explored to determine its productivity in terms of its theoretical maximum sustainable yield the same can be done for a simulated zone. One difference is that by applying a constant fishing mortality rate across the zone the total catch or yield would still need to be distributed in proportion to the available exploitable biomass in each population. The catch is assumed to be distributed by the divers in relation to the available exploitable biomass, see equation (82), and in the usual dynamics of fishing random noise is included to allow for errors in estimates of the cues that indicate availability and future availability. When estimating the potential yield from a zone no random noise is included and the catches are distributed in the ideal proportions relative to the available exploitable biomass in each population. This means that the estimate is a maximum or optimum estimate and that, in reality, errors are likely to degrade the effectiveness or efficiency of fishing.

The process of determining the productivity of a simulated zone is therefore the same as when estimating the productivity of individual populations; thus an array of increasing annual harvest rates are applied to an unexploited zone and the eventual stable yield each harvest rate generates to is noted and the maximum identified. In the case of whole zones, to allow for a longer time to reach approximate equilibrium yields 100 years of fishing were used instead of the 50 that were used for individual populations.

19.6.2 Results

A range of harvest rates from 0.03 to 0.3 were examined which estimates the MSY to be approximately 633 tonnes. This yield is produced once the zone is depleted down to $37.98\%B_0$ and, if catch rates are directly related to exploitable biomass, this would mean that catch rates would likely be only 27% of those in the unfished zone. It also implies that up to 23% of all exploitable biomass could be taken each year on average. A zone

of 70 populations having a BML of 138 mm was the basis of most calculations within the MSE analyses.

Table 32. Predicted yield from years 95 and 100 and the related depletion levels for the spawning and exploitable biomass levels for an array of different harvest rates applied for 100 years to a simulated zone of 70 populations with a BML of 138 mm. The highlighted row relates to the predicted maximum sustainable yield (MSY). Estimates are only approximate as an ideal equilibrium is not reached in any case (compare Catch95 with Catch100).

Harvest	Catch 100	Catch 95	Spawning	Exploitable
0.03	243.550	243.552	0.8380	0.8023
0.04	303.290	303.292	0.7942	0.7494
0.05	354.855	354.859	0.7542	0.7014
0.06	399.388	399.393	0.7178	0.6579
0.07	437.841	437.847	0.6844	0.6182
0.08	471.015	471.023	0.6536	0.5819
0.09	499.591	499.600	0.6253	0.5486
0.10	524.144	524.156	0.5991	0.5180
0.11	545.168	545.182	0.5748	0.4898
0.12	563.087	563.104	0.5522	0.4638
0.13	578.268	578.288	0.5311	0.4396
0.14	591.026	591.050	0.5114	0.4172
0.15	601.637	601.666	0.4930	0.3964
0.16	610.343	610.376	0.4758	0.3770
0.17	617.353	617.392	0.4595	0.3589
0.18	622.852	622.899	0.4443	0.3420
0.19	627.005	627.059	0.4299	0.3261
0.20	629.955	630.018	0.4163	0.3113
0.21	631.829	631.903	0.4035	0.2974
0.22	632.743	632.828	0.3914	0.2842
0.23	632.796	632.894	0.3798	0.2719
0.24	632.080	632.194	0.3689	0.2603
0.25	630.676	630.807	0.3585	0.2493
0.26	628.657	628.807	0.3487	0.2390
0.27	626.089	626.260	0.3393	0.2292
0.28	623.031	623.225	0.3303	0.2199
0.29	619.537	619.757	0.3218	0.2111
0.30	615.655	615.905	0.3136	0.2028

Of course, given natural variation in recruitment and random noise in other factors it is unlikely that such a yield would really be sustainable in the long term. The production curve is skewed to left such that approximately the same yield can be obtained across a relatively wide range of harvest rates and depletion levels (**Figure 83**). Economically it would be most efficient to capture the potential yield using the least effort, which means the lowest harvest rate. Optimizing this, however, would require cooperation among all divers to distribute effort appropriately; at least minimizing overlap of areas fished immediately following a fishing event by one diver.

Because part of the spawning biomass is protected by the LML, while the exploitable biomass is everything larger than the LML, the exploitable biomass will always be less than the spawning biomass.

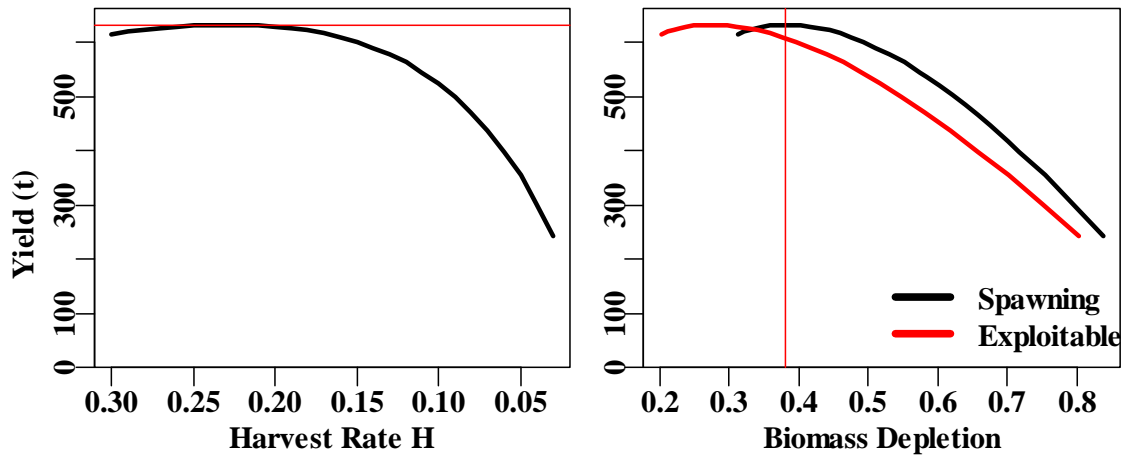


Figure 83. The theoretical productivity of a simulated zone of 70 populations with an overall BML of 138mm. The biomass depletion relative to the unfished state and the straight red lines depict the MSY at 632.8 t.

This theoretical productivity is smaller than the theoretical combined MSY from each of the 70 populations (which in this case is 662.163 t). This is merely a reflection that not all populations have equivalent dynamics and the distribution of catches is not ideal even though it uses the relative distribution of exploitable biomass. The yield produced by the zone is relatively flat across a wide range of harvest rates and consequently of catches and depletion levels.

20 Legal Minimum Lengths and TACs

20.1.1 Preamble

Most of Section 20, which describes a study of the interactions expected between modifications to the Legal Minimum Length and associated Total Allowable Catches in blacklip abalone, has been formally published as:

Haddon, M. and F. Helidoniotis (2013) Legal minimum lengths and the management of abalone fisheries. *Journal of Shellfish Research* **32**: 197-208.

20.2 Introduction

The management of the Tasmanian abalone fishery (95% blacklip abalone, *Haliotis rubra* and 5% greenlip abalone *H. laevis*) uses a number of regulatory instruments (Tarbath & Gardner, 2011). These include 1) divers require an abalone licence (123 available), 2) a Total Allowable Catch (TAC) is allocated as individual transferable quota (ITQ) in five different quota zones around the State plus a separate greenlip quota, 3) to further distribute effort, catch caps are placed on some statistical reporting blocks within zones; these caps are effectively loose competitive TACs, and finally 4) each zone has at least one legal minimum length (LML). In addition, *ad hoc* block closures are sometimes put in place in areas thought to need resting. The spatial management measures (zones, TACs, catch caps, and *ad hoc* closures) aim to distribute the catch in a manner that avoids local depletion while the different LMLs around the State aim to preserve some (unspecified and unknown) level of spawning biomass. All of these management tools are commonly used elsewhere.

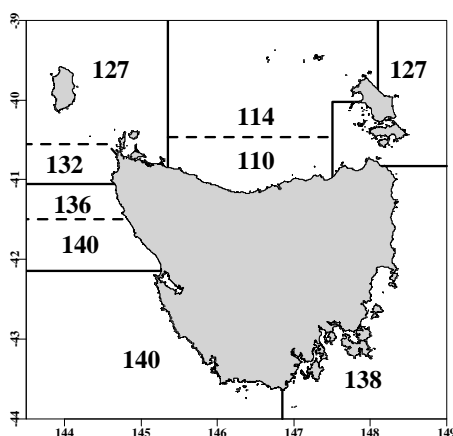


Figure 84. Schematic map of Tasmania with the recent articulation of different Legal Minimum Lengths (LML) in mm for blacklip abalone (*Haliotis rubra*) around Tasmania Australia. Solid lines separate different quota zones while dashed lines indicate LML sub-divisions within zones. Revisions are continually being suggested, with recent attention being paid to the Northeast and Northwest, both of which are in the same quota zone.

In Tasmania, the guideline for setting the LML for a zone is to estimate the average size at maturity and then add two years' growth. Although there are no prescribed methods for how to do this, in practice the average growth increase expected for two years is added to the average size at 50% maturity; the expected growth increments are those predicted by the Inverse Logistic (Haddon et al, 2008). Unfortunately, there is a great

deal of spatial heterogeneity among separate populations of abalone in terms of their biological properties relating to growth and maturity. Thus, any LML selected for relatively large geographical areas (like a quota zone) is always a compromise between providing more than two years protection to the relatively slow growing populations and less than two years protection to the faster growing populations. Because of this problem, despite the so-called two-year rule being included in the Tasmanian abalone management plan (currently under revision), in practice it can only be used as a guideline rather than a regulation.

Initially, in Tasmania, the LML was set State-wide at 127 mm in 1962, but now each zone has its own minimum length regulations; Tarbath & Gardner (2011) provide a detailed history of developments. Even within a zone different areas can have a different LML if the variation of localized abalone populations makes that desirable, although no zone currently has more than two LML (Figure 84).

Because of the compromise nature of the protection afforded different populations by a given LML, suggestions for changing a LML are often controversial among the abalone industry. Increasing a LML will increase the protection of mature biomass, with currently unknown but assumed precautionary consequences, but it will also lead to a reduction in the amount of catch that can be taken from some of the less productive populations. Assuming the TAC is not reduced, increasing a LML will thus lead to greater pressure being placed on more productive populations. This increased pressure may be offset by the larger abalone weighing more, which implies that fewer abalone would be required to make up the same weight of catch. However, if the LML is increased too much then fewer animals will be able to grow into legal sizes, catches will be reduced, and the quality of the product may decline as older individuals are sometimes reported to develop thicker shells, have a lower relative meat yield, and even develop a discoloration of the foot tissues. Discussions about what constitutes the most appropriate LML for different stocks are hampered because it remains unknown what level of mature biomass protection is required to secure a stock's sustainability. It is also unknown what level of protection is provided to the spawning biomass by a given LML, or how changing a LML might affect the potential catch from an area, either as weight or in terms of the number of individuals needed to take a given catch.

A number of abalone fisheries around the World have exhibited significant declines in their production (Hobday *et al.*, 2001; Hamasaki & Kitada, 2008; Neuman *et al.*, 2010; Searcy-Bernal *et al.*, 2010) and while the exact reasons for such declines are varied, an important aspect is a failure to recruit in adequate numbers to maintain the stocks. Being relatively sedentary, the successful reproduction of abalone is sensitive to the effects of severe depletion brought on by fishing, disease, or environmental change (e.g. sea temperature and weed cover). While disease and environmental changes are difficult or not amenable to short-term management, depletion of mature biomass brought about by fishing should be manageable through the combined influence of a LML and a TAC. Trade-offs are expected between the TAC set and the LML selected to avoid depletion of mature biomass. The risk of depletion of the mature biomass is likely to be greater if catches are close to and potentially higher than the maximum productivity of a stock. In such cases, setting a LML that secures a proportion of the mature biomass becomes important for ensuring sustainability. If, however, the TAC is low relative to the productivity, then the LML can be set at a lower size with a lower risk of depleting mature biomass. This trade-off could be useful if smaller sizes commanded premium prices in commercial markets, which has sometimes happened in the Tasmanian live export market; however, the exact character of this trade-off is currently unknown. In terms of con-

tributing to sustainable fishing, because the weight at length relationship is exponential the numbers of abalone required to take a given TAC will be lower with a larger LML; this is also part of the trade-offs that should be considered when setting a LML.

In this present work we use a Management Strategy Evaluation (MSE) simulation framework (Section 19) to examine how different LMLs interact with a typical range of biological variation exhibited within a simulated abalone quota zone and lead to an array of different protection levels across the various populations. In this way we can determine the proportional protection of mature biomass expected from different LMLs and the range of protection levels afforded populations having different productivity. At the same time, the trade-off between the TAC, the numbers required to take the TAC, and the proportion of mature biomass protected by different LMLs, was assessed by determining the TACs that led to the same target level of stock depletion under different prescribed LMLs.

The objectives of this section were therefore:

1. To develop a simulation framework capable of mimicking an abalone fishery made up of numerous populations, each with somewhat different biological properties.
2. Determine the proportion of mature biomass secured by different LMLs in simulated populations with known properties.
3. Determine the trade-off between LML and number of abalone required to take a TAC while inducing the same level of depletion of mature biomass in a simulated fished population.

20.3 Methods

20.3.1 The Simulation Framework

Spatial structuring is a fundamental aspect of an abalone stock and this is included by simulating multiple separate populations (in this case 70) each with their own set of particular biological properties. The choice of 70 separate populations enabled the full range of biological properties to be expressed given the random allocation of values taken from the data used to condition the operating model (Section 19), but the actual number of populations is arbitrary and the MSE framework can simulate as many or as few as wanted. The time step is annual with natural mortality being implemented in two halves with the remaining dynamics in between so as to spread the effects of natural mortality across the implied annual dynamics. Because the emergent and cryptic components are modelled separately the model equations differ from earlier size-structured models (e.g. Breen *et al.*, 2003), particularly in the annual dynamics and in the model initiation (Section 19). The dynamics involve interactions between growth, survivorship and recruitment (as in previous models) but the interactions between the emergent and cryptic components of each population can also be important.

20.3.2 The Impact of Different LML on Yield and Spawning Biomass Protected

Size at maturity values were selected such that zones containing 70 populations could be simulated with Biological Minimum Lengths (BML), averaged over 20 replicate zones, that approximated four chosen sizes, which were 122, 127, 132, or 138 mm. By setting LMLs also of 122, 127, 132, and 138mm for each of the four BML it was possible to consider how the MSY and the proportion of the mature biomass protected by the LML varied across the 16 possible combinations. Naturally, the combinations with the

smaller BMLs (from a smaller size at maturity) generated smaller MSYs because the average maximum size was also smaller; comparisons were thus limited to the four LMLs within each BML.

20.3.3 Conditioning the Simulation Model

While there are detailed and adequate amounts of catch and catch rate data, data on growth and on size at maturity are only available in sufficient quantities to reinforce the intuition from previous studies that abalone stocks are spatially heterogeneous in their biological properties at small spatial scales (Figure 85), potentially as small as tens to hundreds of metres (Prince *et al.*, 1987, Helidoniotis *et al.*, 2011). However, by selecting some of the biological properties at random from the range available (generally characterized with either normal or log-normal distributions), then simple linear and non-linear relationships between variables can be used to determine related parameters in equations describing processes such as growth, size at maturity, and size at emergence. Thus, for example, it was only necessary to select two out of the three parameters describing growth in each population because there was a tightly fitting linear surface relationship found between the three growth parameters across the 27 populations studied for growth characteristics. In an analogous fashion, there was a tightly fitting logarithmic relationship between the two parameters describing the weight at length relationship for 122 separate populations, which could be used to simplify the allocation of this biological property to each simulated population. Not only were there relationships between the parameters of curves describing particular biological properties but there were also relationships between the various biological properties within populations. For example, populations that have a smaller size at maturity also have a smaller maximum length and often a lower weight to shell length relationship (Figure 85). These variables all have an influence on the relative productivity of different populations. Using these relationships simplified the conditioning of the simulation framework onto a particular quota zone using the data available in Tasmania. The equations describing the dynamics and the supporting equations describing growth, size at maturity, weight at length, selectivity, size at emergence and other processes affecting the biology and fishery for each population are detailed in Section 19.

20.3.4 The Number of Abalone Required for a Given Catch

The LML for blacklip abalone on the Tasmanian east coast has previously been 127 mm but is now 138mm, which is a compromise reflecting the BML observed across > 200 east coast populations (unpublished data). To examine the effect of these two LML, relative to the average BML, on the number of abalone required to achieve the same catch and the same level of depletion, comparisons were made of outcomes when LML of 127 and 138 mm were applied to 50 replicate zones that had an average BML of 138 mm. With 70 populations, the total MSY across each of these replicate zones was approximately 750 t (more precisely the values were 756 t at an LML of 127 mm and 751 t at an LML of 138 mm). The simulations applied four different TAC (650, 675, 700, and 750 t) at the two different LML. Before imposing any fishing each replicate zone was depleted to approximately 0.4 B0 by iteratively searching for and applying the fishing mortality which, when repeatedly applied generates the required 0.4 B0. With each pre-depleted replicate zone, the selected TAC was applied in each year for 50 years and in each year the actual catch, the mature biomass, the depletion level across the zone, and the number of abalone taken in the catch are recorded. The median values in each year for these four variables, across the 50 replicates, were then compared across the four TAC levels and the two LML settings.

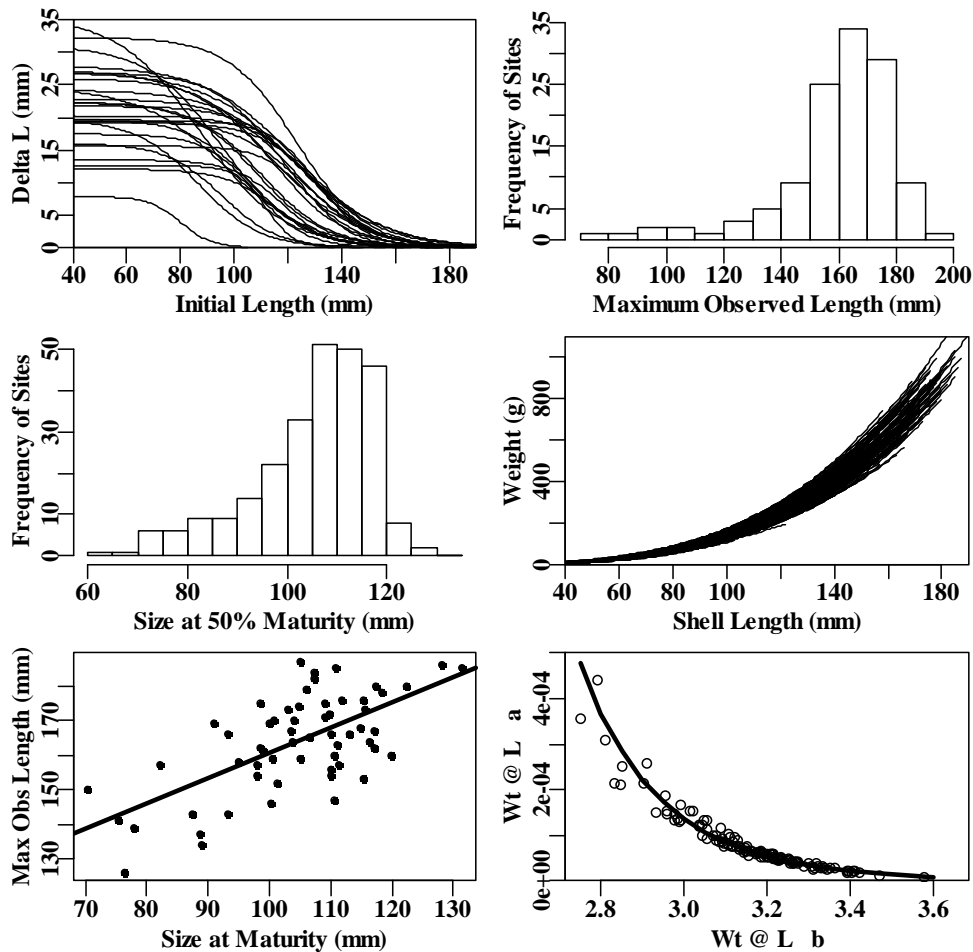


Figure 85. Examples of variation observed in the biological properties of populations of blacklip abalone (*Haliotis rubra*) at different sites around Tasmania. Top left is the spread of the predicted average growth increments for 27 different populations, top right is the observed maximum lengths seen in 122 different sites, middle left is the distribution of the size at 50% maturity at 258 sites east of Longitude 146.5, middle right represents the distribution of the expected weight at length at 122 sites (truncated at the maximum length observed at each site), bottom left is the relationship between size at maturity and maximum observed length (both the relationship and its variability are required), and finally, at bottom right is the relationship between the two parameters of the weight at length relationship.

20.3.5 The Calculation of Mature Biomass Protected

There are numerous ways in which an estimate of the proportion of mature or spawning biomass protected in a given population by a given LML could be calculated. Here we compare the total unfished mature biomass or B_0^{Sp} (see Section 19), with the unfished mature biomass vulnerable to fishing, B_0^M . Observations from the fishery imply that the use of logistic selectivity around the LML is more appropriate than assuming knife-edged selectivity at the LML. However the vulnerable mature biomass is calculated, the proportion, P^{Sp} , of mature biomass protected by a given LML in any year t is:

$$P^{Sp} = (B_0^{Sp} - B_0^M) / B_0^{Sp} \quad (83)$$

This represents the maximum amount of protected spawning biomass as once the stock begins to be depleted (i.e. the subscripted time is no longer 0) this can also lower the amount of spawning biomass below the LML.

20.3.6 Simulating a Quota Zone

A quota zone is defined as a subdivision of the overall fishery that includes multiple populations (in this case, 70 populations), but are managed with the same LML. To explore the effect of different LML values on the level of protection afforded a particular quota zone, replicate simulations consisting of 70 populations were generated. The production curve for each population was characterized numerically by repeatedly applying an array of constant harvest rates (0.01 to 0.4 in steps of 0.01), each for 50 years, to estimate the equilibrium yield and concomitant depletion level at each harvest rate; 50 years was always sufficient to achieve equilibrium. The number of simulations for each population was thus 40 different harvest rates at 10 different LMLs (121 – 127mm, and 130 – 140mm in steps of 2mm) giving a total of 400. Each of the 40 different harvest rates gave rise to an equilibrium yield. The maximum sustainable yield (MSY) across the zone for each LML scenario was thus estimated by finding the maximum equilibrium yield possible from summing the individual equilibrium yields from each of the 70 populations for each harvest rate. This is operating at a population level whereas the finest level of management in the real world operates at a statistical block level, which would, for most blocks, include more than one population. For each population the proportion of the mature biomass protected by each of the given LML scenarios was calculated using equation (83) and the average for the zone was obtained by using the sums of the 70 B_0^{Sp} and B_0^M .

The Biological Minimum Length (BML) is defined as the outcome of applying the two-year rule to each population separately. The curve describing the proportion mature at a given size and the growth characteristics were different for each population so the BML also varied between the 70 populations.

20.3.7 Simulating Catching the TAC

Before further exploring the dynamics relating to alternative LMLs, each zone was depleted to approximately $0.4B_0$ by applying a constant harvest rate until the desired depletion level was approximated. This initial depletion was used to simulate the behaviour of an active fishery rather than always starting with pristine, unfished populations.

The relative productivity of each population was characterized by determining what proportion of the total MSY was generated in each population. Catches were allocated to each population by multiplying the TAC by this vector of proportions and adding a small amount of normal random noise. These individual catches were applied to randomly selected populations. Because of the added noise, however, to prevent the TAC being excessively over- or under-caught, once 25, and again once 50, populations had been exposed to fishing the remaining TAC was reallocated to the remaining unfished populations, which would either increase or decrease their respective catches appropriately. Each simulation consisted of 50 replicates of 70 populations fished for 50 years, and each year the numbers required to take the predicted catch were also estimated. This simulation implies that some populations may become overfished while others are possibly under-fished, which reflects the variation in localized catch levels and assumed depletion levels seen in the Tasmanian fishery (Tarbath and Gardner, 2011).

20.4 Results

20.4.1 Individual Populations

Within individual populations the size at 50% maturity and the BML interact with the growth characteristics to influence the proportional numbers at size and, in conjunction with the LML, the proportion of mature biomass secured from fishing pressure (Figure 86). A larger size-at-maturity correlates to a larger L50 growth parameter, and, even where the other growth parameters remain the same, this leads to abalone growing to a larger and heavier final size. When the LML aligns with the BML then the proportion of the mature biomass protected from fishing tends to be just above ~20% (Figure 86). However, if the LML is set above the BML the protection can become much greater, which would prevent access to a significant proportion of the animals in such populations. Alternatively, if the LML is set too small, then the proportion protected can be relatively small which could lead to sustainability problems (Figure 86).

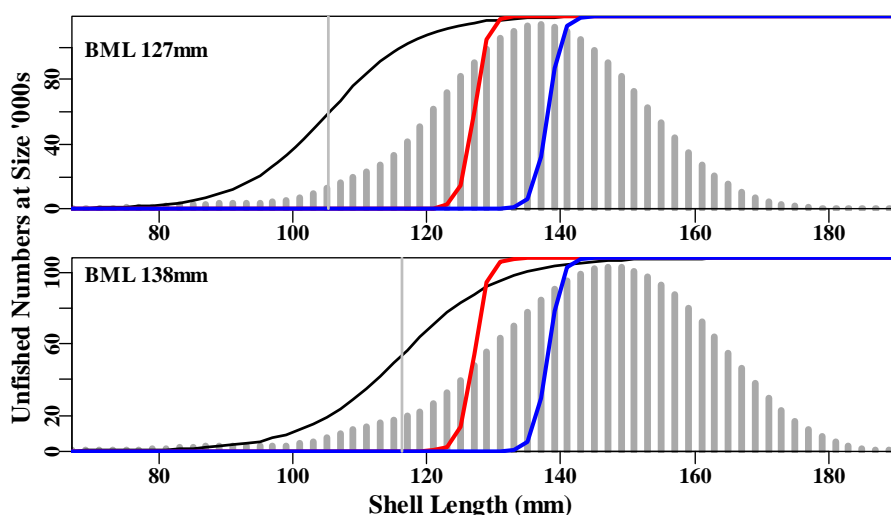


Figure 86. The unfished size distributions of emergent abalone when the size at 50% maturity (the vertical grey line) is 105.3 mm and 116.3 leading to BML values of 127 mm and 138 mm respectively. In each graph the black line is the curve describing the proportion mature at a given size, the left hand selectivity curve is centred around 127mm, and the right hand selectivity curve is around 138 mm. With a BML of 127mm, the mature biomass protected by an LML of 127mm was 22.1% but was 49.9% with an LML of 138mm. With a BML of 138mm, the mature biomass protected in this population by an LML of 127mm was only 7.6% and was 23.4% with an LML of 138mm.

20.4.2 The Range of Protection within Zones

When the size at 50% maturity is set such that the expected BML averaged across 20 simulated zones should be 138 mm the spread of the $\text{Max}\Delta L$ (the initial growth increments), the size at maturity, the maximum length, and the BML are all described by approximately normal distributions. However, the unfished mature biomass, B_0 , and MSY are both distributed in a log-normal fashion (Figure 87). This general pattern was common across each predetermined BML. For the BML of 138mm, the range of maximum yields extended to just above 70 t and the percent protection of mature biomass ranged from 5.5 % to 66.6 %, although the higher levels of protection were mostly limited to those populations with maximum yields less than 10 tonnes.

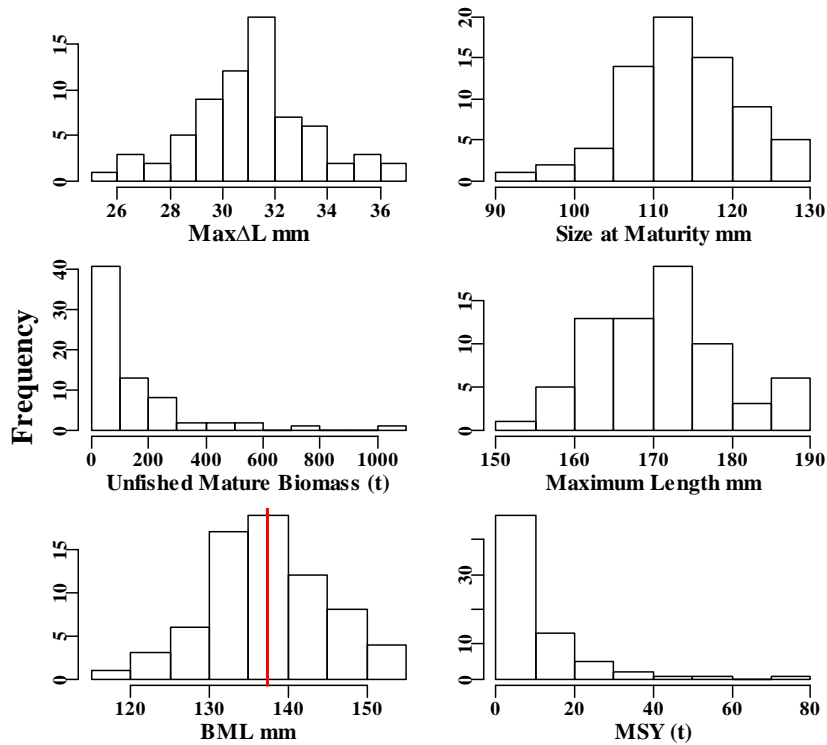


Figure 87. The properties of 70 abalone populations in a simulated zone in which the mean BML across 20 simulated zones is expected to be 138 mm. MaxΔL is one of the parameters describing growth, the maximum length is the length after 23 years of growth, and the BML is the biological minimum length, where the vertical line is the average for this example zone at 137.5mm.

If the percent of mature biomass protected is plotted against the related size at maturity and the MSY for each population the relative protection is distributed unevenly across its range with fewer populations having the smallest size-at-maturity values (Figure 88). Most of the populations with the highest levels of mature biomass protection only have relatively low expected yields. Combinations of high yield and high protection do not appear common (Figure 88), reflecting that the highest yields are likely to derive from populations that grow fastest and largest.

20.4.3 Percent Protection of Mature Biomass with increasing LML

When replicate zones are simulated the variation in the average BML generated in each zone becomes evident and the effect of different LML on the percent protection afforded the mature biomass can be illustrated (Figure 89). In each of the four cases, when the LML matches the average BML approximately 20% of the mature biomass is afforded protection, although a BML of 122mm leads to protection slightly below 20% and that from a 138 mm BML being slightly above. If the LML is set at or more than 10mm below the BML the percent protection of mature biomass declines below 10%. The trends representing the average percent protection for each BML are all approximately parallel to each other (Figure 89). The variation within each zone is much greater than the averages between zones (compare Figure 88 with Figure 89).

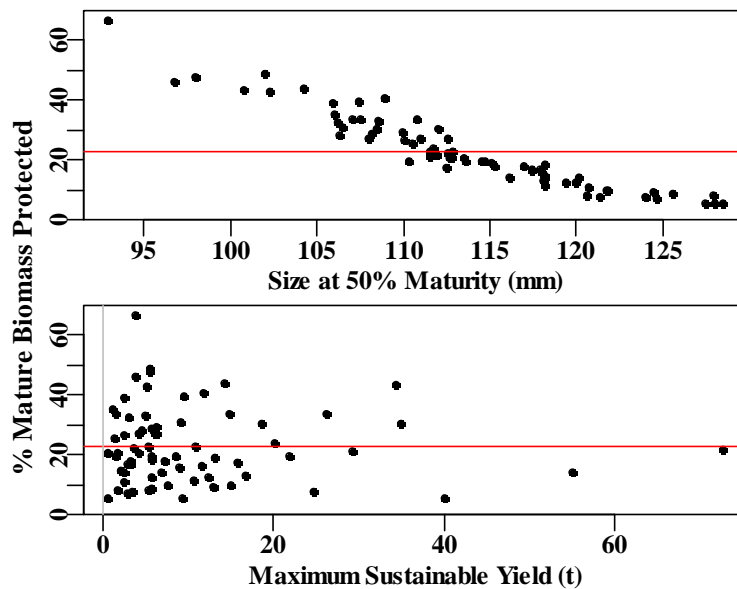


Figure 88. With a BML of 137.5 mm and a LML of 138 mm, the average percent protection of mature biomass was 23.2 % (the horizontal line in both graphs) with a range from 5.5 – 66.6%. The total MSY across the zone was 751.1 t with a range across populations from 0.6 t to 72.6 t.

20.4.4 Variation of MSY with LML

The LML imposed upon replicate zones influences the selectivity of fishing and this in turn influences the average MSY across each set of replicate zones. For each of four average BML scenarios (122, 127, 132, and 138 mm) the MSY varied in a smooth manner depending on the imposed LML (Figure 90). In each of the four cases the maximum MSY is always predicted to occur at a LML which is between 2 – 4 mm smaller than the associated BML (so, the percent protected mature biomass <20%). There is a transition of the curves from a shallow dome shape for the BML of 138 mm, with a maximum at an LML of 134 mm, to a continually declining curve for the BML of 122 mm with its maximum at the lowest LML of 121 mm (Figure 90). The curves for the BML of 132 and 138 mm are relatively flat but slightly domed. In the simulations, with a BML of 138 mm the average MSY at a LML of 138 mm was approximately 5 tonnes less than at a LML of 127 mm (< 1% difference).

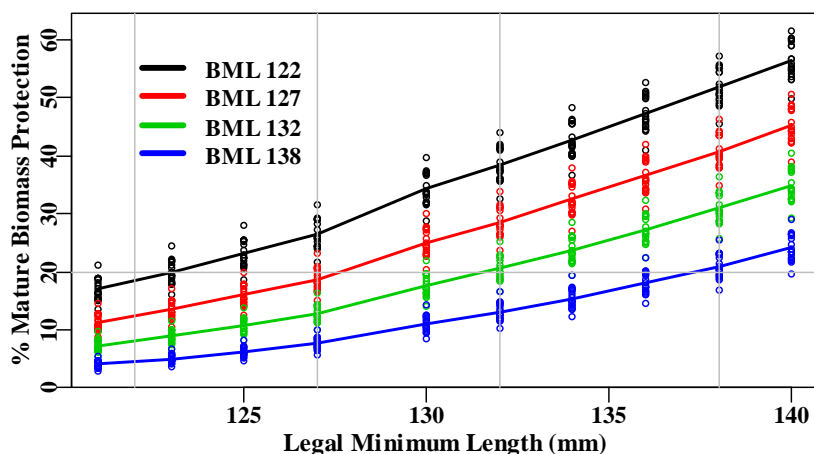


Figure 89. The percent protection afforded mature biomass at 10 different LML for four different average BML values. For each combination of BML and LML, the lines in each case are the average protection level, while the data points are the variation obtained in each of 20 replicate zones. The fine horizontal line is at 20% protection while the vertical lines are at 122, 127, 132, and 138 mm. The top line and data points relate to the BML of 122 mm while the bottom line and points relate to the BML of 138 mm.

However, for the BML 138 mm case, given that the overall mean across all replicate zones and different LML was 726 t and the range across individual zones was 558 – 915, a difference of five tonnes is not significant. A similar argument could be made for the BML 132 mm case, however, for the BML 127 mm case while an LML of 121mm suggests a MSY only two tonnes greater, a LML of 132 mm would lower the expected TAC by 20 tonnes (3.4%) and a LML of 138 mm would lower the TAC by 73 t (12.1%). For populations with smaller BMLs, imposing a smaller LML than the BML affects the MSY far less than imposing a larger LML (Figure 90).

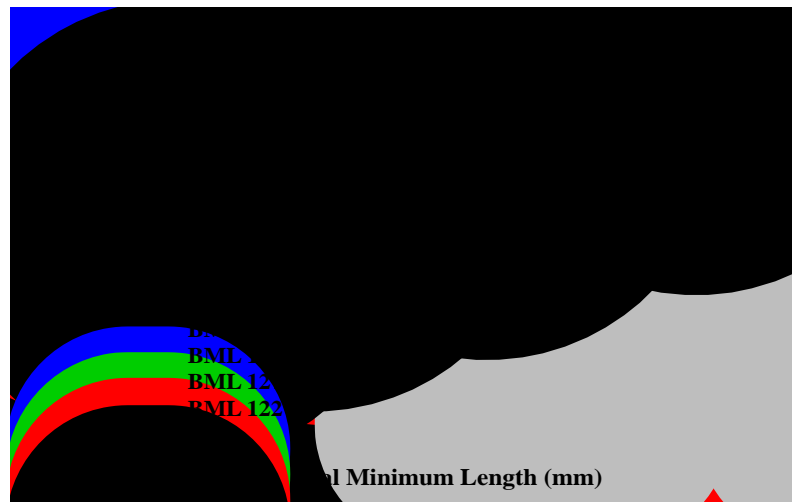


Figure 90. The manner in which the average MSY across 20 replicate zones changes relative to 40 combinations of four BML (122, 127, 132, and 138 mm) and 10 LML (121, 123, 125, 127, 130, 132, 134, 136, 138, and 140 mm). The maximum size and weight of abalone are influenced by the BML so the absolute MSY for the BML of 122 mm was naturally smaller than that generated with a BML of 138 mm. The circled points are where the average BML matches the LML; these are never at the maxima of each curve.

20.4.5 Changes in Numbers of Abalone in the Catch with LML

When the LML is smaller the fishery can take smaller and lighter abalone so the expectation is that for a given zone the number of animals needed to land a given TAC will be greater for a LML of 127 mm than for a LML of 138 mm. In terms of the percentage increase in numbers of abalone required when fulfilling the TAC at 127 mm rather than 138 mm, both the average increase in the percentage and the spread of the increases required become larger as the TAC increases; in all cases an average increase of more than 10% is required (Figure 91). Given that the numbers required for a TAC between 650 and 750 t are between 1,360,000 and 1,830,000 the differences between using a LML of 127 mm and 138 mm can vary between 148,000 and 236,000 animals depending on the TAC imposed (**Table 33**).

In this simulation the percent protection afforded the mature biomass by the two different LML of 127 and 138 mm was 8.1% and 22.2% respectively. Applying a different given TAC for 50 years led to different final depletion levels for the two treatments. To obtain the same depletion level required taking a smaller TAC with the LML 127 mm. The simulated zones had an average total MSY of about 750 t and applying a TAC of 750 t with a LML of 138 mm led to the median depletion after 50 years approaching a

new equilibrium of approximately 30% B_0 while a TAC of 675 t led to the zones remaining at approximately 40% B_0 . With a LML of 127mm a TAC of 650 t was required to achieve approximate stability but a TAC of 750 t led to the median depletion across the replicated zones declining steadily down to approximately 22% B_0 with no indication of the stock reaching equilibrium (Figure 92; **Table 33**). To obtain a depletion level of 40% B_0 after 50 years of fishing at 127 mm LML one needs a TAC of 650 but at an LML of 138 mm a TAC of 675 t can be taken. As the TAC allocated gets closer to the total MSY then the difference between the two LMLs to get the same depletion level increases. Thus to finish at approximately 30% depletion a TAC of about 700 t and 750 t is required by the 127 and 138 mm LML respectively (Figure 92; Table 33).

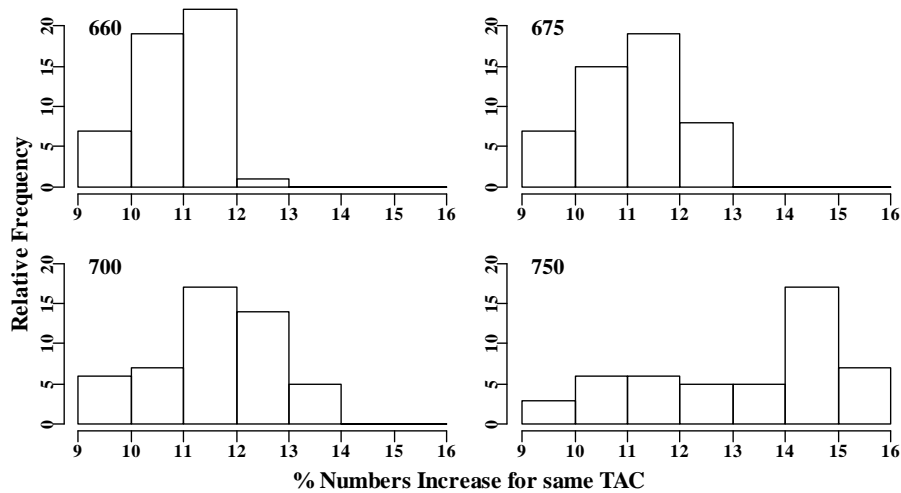


Figure 91. The average increase in the percent of numbers needed to catch the same TAC when the BML = 138 and the LML is 138mm relative to a LML of 127 mm. The numbers at the top left of each graph is the TAC applied. There were 50 replicate zones in each case.

Table 33. The numbers of abalone required to take a given TAC when the BML is 138 mm under two different LML. % Increase is the average percentage increase in numbers, across 50 replicate zones and 50 years, required to take the same TAC under an LML of 127 rather than an LML of 138 mm. All simulated zones started at an initial Depletion level very close to $0.4B_0$. The Depletion columns are the median stock level after 50 years of applying the TAC and the Numbers ‘000s are the median numbers required to take the TAC in the 50th year of fishing.

TAC	% Increase	LML 127 mm		LML 138 mm	
		Depletion	Numbers ‘000s	Depletion	Numbers ‘000s
650	10.5	0.401	1509	0.432	1361
675	10.6	0.357	1603	0.403	1428
700	11.1	0.332	1688	0.352	1492
750	12.7	0.224	1832	0.315	1596

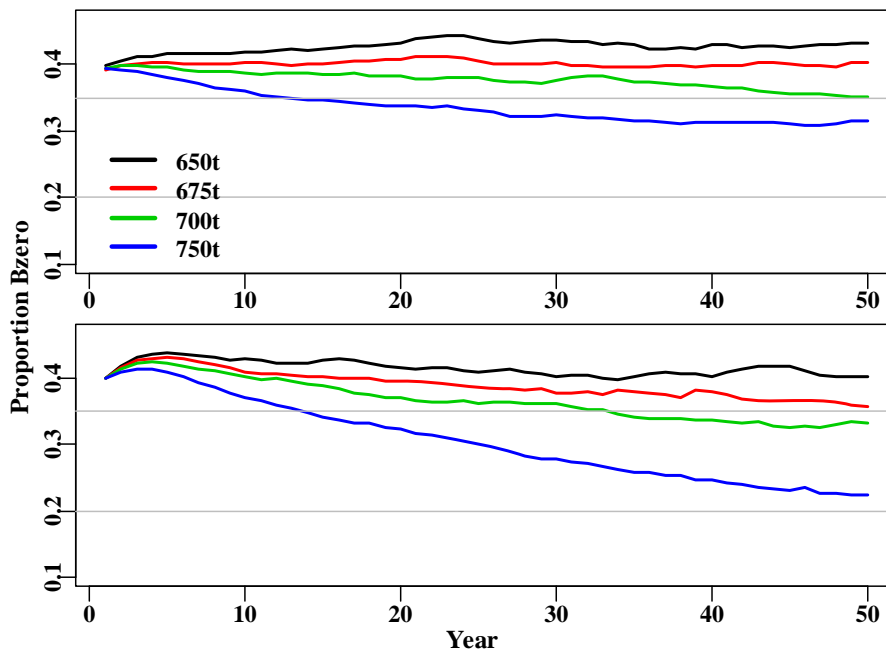


Figure 92. Different responses of a zone with a BML of 138 mm leading to an average total MSY ~ 750 t depleted to approximately $0.40 B_0$ and then exposed to 50 years of four different TAC levels. The top graph illustrates the imposition of an LML of 138 mm while the bottom graph indicates the response under a LML of 127 mm. The two horizontal grey lines are at 0.20 and $0.35 B_0$. Note the y-axis begins at $0.1 B_0$. The top line in both cases relates to a TAC of 650 t and the bottom line a TAC of 750 t.

Fishing at a LML of 127 mm when the BML is 138 mm greatly affects the size distribution of emergent abalone. By summing the size distributions of emergent abalone across the 70 populations in either case the resulting size distributions after 50 years of fishing at 750 t are very different (Figure 93). The size distribution when fishing at a LML of 138 mm is relatively steep from about 138 mm up to about 165 mm and the mode of legal animals is highest at the LML. Fishing at a LML of 127 mm, the size distribution is rather flatter between 127 mm and about 165 mm but the mode of the fished animals is at 130 mm rather than 127 mm (Figure 93). The range of the size distribution when the stock is at $40\% B_0$ is wider when fishing at a 127 mm LML relative to fishing at a LML of 138 mm, ranging from 127 to about 181 mm relative to 138 to about 170 mm.

20.5 Discussion

This appears to be the first time that a size-structured simulation model has been used to examine interactions between the LML, the TAC, the percent mature biomass protection, and the number of animals needed to take a given catch. The use of such a simulation is the only plausible way to examine the trade-offs between the objectives of maximizing the TAC, maximizing the proportion of the mature biomass protected, and minimizing the numbers of abalone required to take a given catch.

The origin of the two-year rule used for setting the LML in Tasmania appears to have been simple intuition, although in Tasmania essentially the same rule (allowing two years of significant spawning) is used to set the minimum size for scallops (*Pecten fumatus*). In Australia, Tasmania appears to be the only State that attempts to make an explicit relationship between the size at maturity and the LML imposed on particular abalone fisheries.

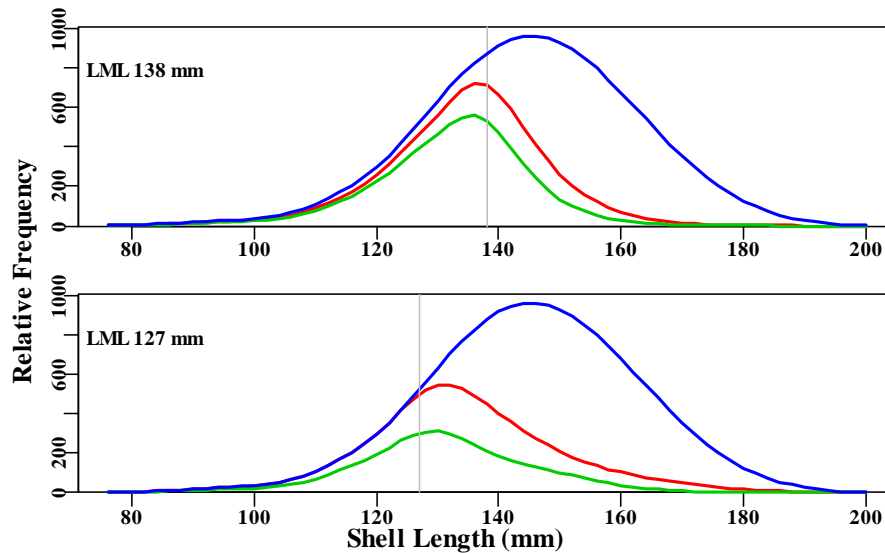


Figure 93. The expected size distribution of emergent abalone with an average BML of 138 mm and a total MSY of 751 t, fished at 750 t. In each case the outer curve is the unfished size distribution, the next curve is the population depleted to 40% B_0 , and the bottom curve is after 50 years fishing at 750 t, leading to depletion levels of 31.5% and 22.4% B_0 for the 138 and 127 mm LML respectively. The depletion level sums across both sizes below the LML and any residual biomass above the LML.

20.5.1 Individual Populations

The maximum yield per recruit (an equilibrium concept) for blacklip abalone appears to occur at sizes a few millimetres smaller than the BML, as evidenced by the maximum MSY for a particular BML always occurring with a LML between 2 and 4 mm smaller than the BML. For the most highly productive populations if the LML is approximately close to the BML there is little difference in the potential yield and the proportion of the mature biomass protected brought about by small adjustments in the precise LML implemented. However, for less productive populations (with smaller BMLs), changes in potential yield and proportion of mature biomass protected were more sensitive to increases in LML. This reflects the continued debate in Tasmania about setting LML values in the north of the State where both growth and size at maturity tend to be expressed at smaller sizes and is also far more variable than further south leading to greater difficulty in finding a compromise LML that is effective across large geographical areas.

20.5.2 Individual Zones

The relationship between the proportion of mature (spawning) biomass protected, the BML (size at maturity plus two year's growth), and the LML imposed is a simple rising curve (**Figure 89**). It suggests that the use of the two year rule is a reasonable compromise when adopting a wide ranging LML. When the LML approximates the average BML then it prevents excessive under- and over-protection within a zone. However, if a zone is fished using a LML that is 5 – 10 mm smaller than the BML then there is a much greater risk of depletion for any given TAC than if fishing closer to the BML. Conversely, if the LML is set 5 – 10 mm above the BML then, especially for populations with smaller BML, there can be very great over-protection such that significant losses in potential yield will occur (**Figure 90**).

It should be noted that the time taken for depletion to occur can be quite extended. In the simulations, dropping whole zones from about $0.4B_0$ to about $0.2B_0$ could take 50

years. However, in that time, some of the component populations within the zone had become so depleted that commercial fishing was no longer viable. If the fishing were occurring at levels well above the sustainable yields then depletion times could be much shorter.

Given the variation typically observed within a quota zone it is the case that a single LML constitutes a compromise between under-protecting some populations and over-protecting others. However, populations that are highly productive as well as being over-protected appear to be uncommon. This is understandable as most highly productive populations are only highly productive in absolute terms because their members grow large and heavy relatively quickly. Very highly protected populations would undoubtedly be under-utilized, however, to be so over-protected implies that their size at maturity must be small relative to many other populations within the zone and thus it is highly unlikely they would be very productive. Nevertheless, individual divers may have localities with which they are familiar made effectively unavailable given an increase in LML, so controversy will undoubtedly remain. The dome shape of the MSY to LML curves for the higher BML values (132 and 138 mm) indicate that smaller LML in such circumstances may lead to growth overfishing. Increasing the LML in such circumstance may lead to increased yields from such populations which would mitigate any increased impacts expected on the more highly productive populations.

20.5.3 Variation of MSY and Numbers of Abalone caught with LML

The intuition that increasing a LML would lead to a serious decline in the available yield from a zone appears valid when a zone is already being fished close to its average BML. However, if a zone was being fished with a LML well below its average BML (for example, Tasmania's east coast has an average BML of approximately 138 mm but was originally fished at 127 mm) then potential yields would not decline to any large extent by moving the LML closer to the BML. An additional advantage would be that the number of animals required to take an equivalent TAC would be expected to decline by at least 10% because they become larger and heavier. Despite this decrease in the number of abalone harvested the larger LML (127 – 138 mm) decreases the exploitable biomass by more than 10% so that the fishing mortality rate will automatically be larger (the same TAC from a smaller exploitable biomass equals a higher harvest or fishing mortality rate). This does not automatically imply a lower catch rate because the catchability of the larger abalone may be rather different from the relatively smaller animals. But, because of this effect large changes should not be made rapidly to LML values so as to avoid rapid changes in the fishery.

In the north of Tasmania, where BMLs can be much smaller, it seems likely that there remain regions in which the LML is inappropriately high and this is preventing the reasonable harvest of available resources. Experimental trials using smaller LML have been used in the northwest in defined regions and discussions and revisions of LML boundaries and levels are on-going.

The expected difference in the size distribution of legal sized abalone under two different LML is large. When the zone is depleted to approximately $0.4B_0$ the size distribution under a LML of 127mm is wider than under a LML of 138 mm (Figure 93). This implies that the fishery under a LML of 138 mm would be more dependent upon new recruits than when under a LML of 127 mm. The size distribution available for capture is another one of the trade-offs that should be considered when selecting a particular LML for a zone. If the size distribution found with the 127 mm LML is preferred then a

more conservative TAC would be required to counter-balance the reduction in protection of the mature biomass so that the impact of the TAC on depletion levels remained stable. Currently it is still unknown what level of spawning biomass to protect using a LML to insure against a stock collapse.

20.5.4 Protection of Mature Biomass

The east coast of Tasmania was fished at 127 mm from 1962 to 1987 after which the state-wide LML was increased to 132 mm, this was increased in the eastern zone to 136mm in 2002 and 138 mm at the end of 2006. This means that for decades many populations on the east coast only had relatively minor mature biomass protection. These circumstances make such populations vulnerable to failures in recruitment success. If severe depletion of under-protected populations coincides with one or two years of negligible recruitment then the spawning biomass could be reduced to very low levels leading to effective collapse for these populations. This is one potential explanation for why some highly productive populations on the east coast went from eight years of landings greater than 100 t with a maximum of almost 300 tonnes during the late 1970s and early 1980s, down to just over 10 tonnes a year following such catches. There are many instances of abalone fisheries around the world where very high catches are followed by collapse and greatly reduced yields which fail to recover. Even a small LML may give the impression that it would provide a minimum secure breeding stock, however, the self-sustaining nature of reproduction in abalone populations and the vagaries of successful recruitment, suggest that LML values which are relatively close to the size at maturity are inherently risky. While many years may go by without apparent problems, if a depleted state coincides with years of weak or no successful recruitment, then particular populations could decline to very low levels from which they may take decades to recover if ever. This would be a particular danger if fishing were proceeding at higher than sustainable levels. The simulations also suggest that slow population declines could occur over decades, even without recruitment failure, which adds a further risk to maintaining a LML which is smaller than the BML. Setting the LML close to the average BML for an area does not risk the loss of significant amounts of yield but offers the potential for reducing the risks of stock collapse, particularly in the face of adverse recruitment events. It can be recommended that any LML chosen should take into account the average size at maturity leaving a significant buffer before permitting fishing mortality (something like the two year rule in Tasmania). If growth data are not available for some areas, some proxy, perhaps related to some fraction of the maximum observable size, could potentially be used instead.

21 MSE of Harvest Control Rules

21.1 Introduction

21.1.1 Objectives, Decision Rules, and Performance Measures

Each combination of data collected, assessment used, and their related harvest control rule or Decision Rule (HCR), makes up a separate management or harvest strategy (Figure 94). The first two elements effectively encompass the estimation of the performance measures (PMs) used. The PMs used form the foundation for whatever HCR is to be used. Some HCRs simply generate a response to the value of the PM, while many include a limit reference point (LRP) and, usually, also a target reference point (TRP); in some the TRP can appear to be informal, for example, where the target is to be anywhere above the LRP. The HCR/Decision Rules operate to manage the fishery away from the LRP (in a positive direction for the stock involved) and, if there is one, towards the TRP. If changes are made to any of the types of data, the performance measures, the assessment used, or the HCR used, then, by definition, we would be using a different management strategy. This is mainly of interest if one attempts to compare an array of alternative management strategies, as is occurring in the present work.

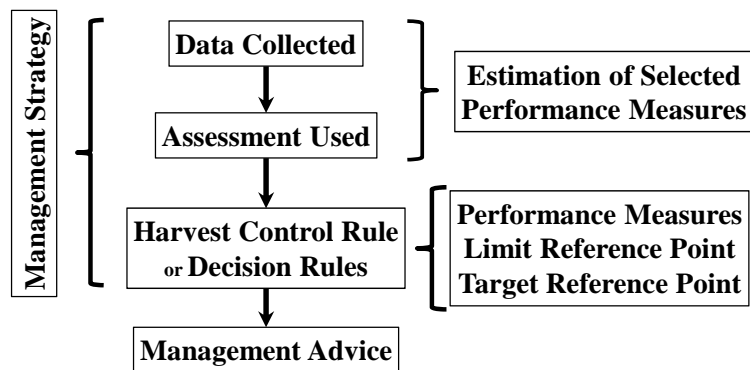


Figure 94. The structure of formal management or harvest strategies. Harvest control rules relate to given performance measures, they often have a limit reference point, and usually have a target reference point. Each combination of data, assessment, and harvest control rule constitutes a management strategy.

The first part of the research presented in this report focussed on trying to identify alternative possible performance measures (PMs) that might be of use in managing abalone fisheries. However, before launching into a detailed examination of PMs different to those currently used, it is worthwhile examining the behaviour of PMs currently used, such as catches, catch rates, and length frequency distributions of landed catches. These PMs have been instrumental in helping to maintain various large abalone fisheries in Australia for 50 years or more (Mayfield *et al*, 2012). Perhaps more important than the PMs used are the Harvest Control Rules (or set of Decision Rules) in which the PMs are used. The very best PM possible will fail to provide good management advice if embedded within an ineffective HCR or a poor set of Decision Rules. It appears to be a reasonable strategy to first optimize the use of PMs currently used to manage Australian abalone fisheries by examining how they interact with different HCR and only then add different PMs and HCRs.

The main driver behind selecting among management strategies (and hence HCR and PMs) is the fishery management objective(s) that managers are aiming to achieve. Such objectives should ideally be set by policy makers and most policies for abalone fisheries within the Australian States and Territories remain generic and loosely aligned with achieving B_{MSY} . As they stand most policies are insufficiently characterized to allow the design of an operational management strategy. In practice, measuring the performance of a fishery relative to B_{MSY} requires a length-based population dynamic assessment model, and is more challenging when assessments are limited to use of empirical performance measures either informally or within formal management strategies.

Currently, for example, the management objective for the Tasmanian abalone fishery remains informal and could be characterized as being one of maximizing the year to year landings while maintaining a sustainable stock; the resulting management actions are reactive rather than strategic. Decisions about each following year's TAC are developed through a series of meetings held during each year during which semi-quantitative discussions are held over any trends in catch rates and in the spatial distribution of catches, with an occasional consideration of the length frequency of catches. Much of the discussion remains qualitative and only relatively informally related to available evidence. The defensibility of any decisions is thus weaker than it could be if the relationship between decision and evidence were to be made formally. In South Australia there has been a recent move towards developing a set of formal decision rules for making decisions about TACs and related management decisions, and a similar approach is in the process of being developed in Tasmania. Such a development would be a positive step towards improving the defensibility of any management decisions. This assumes the management strategies adopted operated successfully and the purpose of the simulation testing here is to determine whether there are any unintended and negative consequences of the approaches being proposed.

Despite the potential improvement in defensibility that developing a formal HCR would engender, without a formal specific objective or specific target for the fishery the success of any particular management strategy will not be able to be assessed prior to implementation in the fishery. With regard to the Tasmanian east coast there have been suggestions about trying to aim for a certain level of productivity, for example, aim to produce 1,000t consistently from the eastern zone. However, such an objective ignores the dynamic nature of any fished stock, 1,000t could certainly be used as an upper limit so that catches were never allowed to increase beyond that level, but it cannot be expected that some such aspirational value will always be a sustainable catch, or be achieved without additional compromise such as lower catch rates. Any objective based purely on a target catch would make it difficult to manage a variable stock sustainably. Effective control only arises from systems that incorporate negative feedback, and the most suitable response to not catching a hoped for catch might be to reduce catches further thereby guaranteeing not meeting the target the following year.

The fundamental objective, at least within the Tasmanian abalone fishery, appears to be to have an on-going fishery while maximizing catch but how to translate that into management actions each year is unclear because what is required to maintain the sustainability of each stock remains undefined. If empirical HCR are developed (as implemented in South Australia and as proposed in Tasmania) then the underlying requirements for sustainability would remain undefined. In the absence of knowledge of the underlying dynamics there has been an on-going evolution of management regulations based upon intuitions. Thus, for example, the legal minimum length (LML) was 127 mm around Tasmania from 1965 to 1986 and the increase to 132mm state-wide in 1987 was

known to protect more of the mature biomass from exploitation and was thus assumed to be a positive move to ensure increased resilience. This occurred at the same time as large reductions in TAC all brought about by a universally acknowledged depletion of the whole fishery up to that point.

As often happens with management measures when they occur together it becomes more difficult to discern what contribution the reduction in total catches and the change in the LML had on the following recovery of the stock that happened over the next 12 years. While it should be possible to use the MSE simulation framework to distinguish between the possible influences of catches and LML this still doesn't define an objective towards which to manage the fishery. It would appear that one of the best things the abalone fisheries in Australia could do would be to put effort into defining an overall objective for each fishery; this would have the added benefit of permitting the success, or otherwise, of management to be assessed and presumably, therefore, to be improved.

21.1.2 Harvest Control Rules/Decision Rules

The phrases 'Harvest Control Rule' and 'Decision Rule' relate to the same underlying idea. The key aspect of the structure of a HCR is that it is a formally defined way to translate one or more performance measures into management advice. A HCR may constitute a mathematical relationship between a performance measure, such as the mean catch rates over the last four years, and a recommended TAC (Figure 95; Haddon, 2012). Or it may be a relatively simple list of rules that lead to a reaction to the present state of affairs, as described by the PMs used, in such a way as to identify the next appropriate TAC (e.g. Dichmont and Brown, 2010).

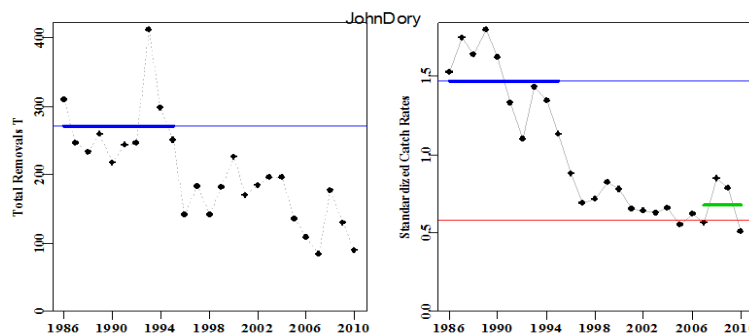


Figure 95. The application of a catch rate based HCR to John Dory (*Zeus faber*) in the South East Trawl fishery. Total removals (catches plus discards) are in the left hand panel while catch rates, with the blue target and red limit reference points are in the right hand panel. The green line represents the mean catch rate from the last four years, which is only just above the LRP and the recommended catch from this analysis reflects that. (Figure modified from Haddon, 2012).

Clearly the objective of management for a particular fishery is vital for determining what sort of management strategy to implement. If the objective were to maximize profitability by managing a fishery so that the average level of spawning biomass was at a level that led to higher catch rates than would be produced if the maximum yield was to be taken (for example 48% unfished spawning biomass, as in Australian Commonwealth fisheries; DAFF, 2007), then a management strategy whose HCR only used the gradient of catch rates (see later) to recommend future catches would not be capable of meeting that objective. Where a HCR could produce and maintain sustainable catches (an aim which is often part of any worthwhile objective), it would be failing to meet the

specific objective of achieving $48\%B_0$. However, where a HCR was based on an empirical consideration of a PM such as catch rates alone, if it had a specific target reference point then some kind of proxy between the HCR's suggested target and the spawning biomass target could be suggested and in this indirect manner the objective could be claimed to have been achieved (Haddon, 2012). Such a specific proxy is not essential, it may be sufficient if the managers and Industry determine that the selected target (assuming the HCR used can achieve it) keeps the fishery in a desirable state. A HCR that only uses the gradient of catch rates (see later) is an example of a HCR that does not have a specific limit or target reference point. Adding a lower limit to the TACs deriving from such a HCR is not the same as putting in place a LRP, all it does is affect the output of the HCR. In simple sets of rules including such a limit is best considered as simply adding another decision rule to the total set so as to get the desired outcome.

21.1.3 The Focus on Decision Rules/Harvest Control Rules

There are numerous PMs available and each PM can also be included in a large array of alternative HCR. In this work we will be considering only the few performance measures that reflect current usage, and these include time series of catches and catch rates. Even with just catch rates and catches there are a very large number of alternative harvest control rules (HCRs) possible and it is these HCR that will be the primary focus of the simulation testing in this section.

One of the problems with the implementation and application of MSE testing to alternative management strategies is that, when the true state of the fishery is unknown the initial range of options is enormous and selecting which to work on is not simple. It would be reasonable to begin by focussing on the management mechanisms currently used in a fishery. In Tasmania the current, relatively informal processes involved each year in determining management advice concerning TACs includes an examination of catch rates which can be detailed at the spatial level of statistical reporting block or sometimes sub-block, within each zone. However, this examination, even where it uses standardized catch rates remains mainly qualitative with respect to any trends, the scale of any changes, and the degree of response to make relative to any observed changes. Instead, we will attempt to translate some of the informal approaches into rules more amenable to inclusion in the simulations. In addition, we will begin with the implementation of the more formal type of decision rules being developed in Tasmania now in 2012. Despite this restriction of focus there remain a very large range of options so we will define a smaller array of alternatives and focus on those.

21.1.4 The Objective of MSE Testing

The primary objective of testing and comparing different management strategies is to determine which is best at providing useful management advice in the face of noisy or poor quality data, or in the face of uncertainty concerning the dynamics underlying the fishery. Criteria for determining whether management advice has been useful would include such things as an ability to manage the fishery in such a fashion that it is able to achieve the explicit objective or even target of the management strategy. In this work we are going to be focussed on the relative performance of different HCR that use catches and catch rates so some more specific criteria for distinguishing the relative abilities of the different HCR are required. These relate to the array of scenarios under which the different HCRs will be tested but it is clearly desirable that for a HCR to be deemed acceptable it should be able to safely manage a fishery which starts in either a depleted state, a slightly depleted state (or close to target state), or even an un-depleted state. In addition, given the natural variability of abalone populations and the fisheries

based upon them, the management strategy that led to the most stable fishery outcomes would be preferred if all other outputs were the same.

As usual with fishery management objectives, trade-offs are to be expected between different desirable management outcomes. All the MSE can do is illustrate these trade-offs it cannot determine the policy that would identify which of any trade-off arrangements would be preferable in a particular fishery. For example, the simulations cannot determine whether it is better to have a more stable fishery at a lower long term catch or a variable fishery yielding a higher long term catch, that would be a matter for policy and should be reflected in the overall objective for the fishery. The simulations may assist in formulating such objectives but only through identifying the range of behaviours that can be traded off against each other.

21.2 The Harvest Control Rules to be Compared

21.2.1 Introduction

Harvest Control Rules (HCRs) do what their name suggests. They constitute a rule or set of rules, which use at least one performance measure (PM) from a fishery to generate recommendations about what catch or harvest to take in subsequent years. There can be many alternative HCRs, even relating to the same PM, such as CPUE. For example, one can relate to some recent estimate of catch rate and recent catch in comparison to some selected target catch rate and from that determine what catch would be likely to move the fishery towards the target catch rate and keep it away from the limit catch rate (Haddon, 2012). Alternatively, it is possible to estimate the gradient of any proportional change in recent catch rates and use that as a guide as to what to do with catches, increasing catches with positive gradients and decreasing them with negative gradients. Each of these HCR can have additional rules added, for example, it is possible that a minimum TAC is required to prevent the infra-structure relating to the fishery from disappearing and for maintaining markets. Unless there is some kind of external impact, such as the occurrence of extensive disease or a large scale natural mortality event, then as long as management has been limiting catches to historical levels there should be no reason to completely shut down a fishery.

Both disease and natural mortality events have occurred in Australian abalone fisheries. The most recent such event was the remarkable marine heat wave with rapid increases in temperature by 5°C from 23°C to 28°C that occurred in Western Australia in February/March 2011. This killed off a very large proportion of the Roe's abalone (*Haliotis roei*) in northern Western Australia and the impact is reported as so great that there was no fishery left (Norwood, 2013).

21.2.2 Catch Rate Gradient Methods

In an attempt to approximate the relatively informal use of changes in catch rates through time in current abalone assessments, we will be examining variants of a HCR that uses the gradient of catch rates across a set number of recent years and using the resulting gradient in some given relationship with TAC change. Variants could use different numbers of years over which to calculate the gradient of catch rates and different relationships between the gradient and degree of TAC change; the data requirements for such a HCR are time series of catch and effort.

Given a time series of catch rates the gradient across the most recent n years would be estimated by first calculating the percent change in catch rates, (ΔCE_y) between years

(this step is required so as to be dealing with proportional changes rather than absolute changes):

$$\Delta CE_y = CE_y / CE_1 \quad (84)$$

where CE_1 is the first year of the group of n years over which the gradient is being estimated, then, for the last n years of available data, solve the linear regression:

$$\Delta CE_y = Inter + Grad .x_{1..n} \quad (85)$$

where x is a simple series $1..n$, where n is the number of years across which the gradient is to be estimated (the default used was eight years but alternatives of 5 and 10 were also considered). The gradient of this regression represents the average proportional change in each year over the period of years selected and so, in order that the total change over the period is accounted for, this gradient is multiplied by half the number of years in the period:

$$\text{multiplier} = 1 + (\text{gradient} \times (\text{period}/2)) \quad (86)$$

where the period is the number of years over which the gradient of the proportional changes in catch rate is estimated. The period is halved because preliminary trials indicated that without the halving the HCR led to highly unstable results. The TAC multiplier is then used to modify the TAC for subsequent years.

21.2.2.1 Gradient HCR Modifier

From experience with age-structured models, which, under some depletion circumstances, found that a catch rate gradient based method led to management advice that encouraged the status quo even when that was sub-optimal, a variant on the gradient HCR was to include a sharp drop in the TAC in the first year of the introduction of this HCR. The intention of this sharp drop in TAC is to introduce some contrast into the catch rate time series as well as initiate a potential rebuild if required. In this case the original TAC at the start of introducing the HCR (which, in the simulations, is done between years 10 and 11) is multiplied by 0.75 to reduce catches by 25%.

$$TAC_{11} = TAC_{10} \times (1 - pert) \quad (87)$$

where TAC_{10} is the TAC in year 10 and $pert$ is the perturbation to the TAC, for example for a reduction of 25% $pert$ would be 0.25. Adding in this initial TAC modifier means this is a different HCR to the unmodified CPUE gradient HCR and it might behave in a very different manner under the range of scenarios considered. The perturbation is only used once at the introduction of the HCR.

21.2.3 Catch Rate Target Methods

Instead of simply changing the TAC in response to the recent catch rates, as in the catch rate gradient method, it is also possible to set a target catch rate towards which the fishery wishes to aspire. The objective of a HCR that would work in this way would be to manage a fishery to achieve the set target catch rate by modifying the TAC up or down depending on a comparison of the current catch rate with the target.

Currently under development in Tasmania and already in use in South Australia is a HCR that can combine an array of different performance measures (PMs) to generate a score and that score is translated into a recommended management action with respect

to the TAC. This approach has been termed a Multi-Criterion Decision Analysis (an MCDA). In the cases examined here, only one empirical PM, CPUE, is used, with a defined selected target CPUE, which enables a detailed dissection of its particular properties.

The HCR consists of two stages: the first is an algorithm that produces a score for each fishing unit (be that a statistical block or a zone) and secondly, the score produced is translated into a TAC modifier which can increase it, decrease it, or leave it as it is. The option selected for examination here involved generating a score for each statistical block or SAU (spatial assessment unit), and then combining those scores to produce a zone wide score. This SAU-derived zone wide score was then used to produce the management recommendation.

The catch rate across populations for each SAU was generated by weighting the catch rates from each of the populations with the proportion of that SAU's total catch and then summing the results:

$$pC_{pop} = \frac{C_{pop}}{\sum_{pop=1}^{n_{pop}} C_{pop}} \quad (88)$$

where pop is each of the n_{pop} separate populations within a given SAU, C_{pop} is the catch taken in a given year within a particular population within a SAU, and pC_{pop} is the proportion of the total catch within a SAU taken in population pop .

$$CE_{Blk} = \sum_{pop=1}^{n_{pop}} CE_{pop} \times pC_{pop} \quad (89)$$

Where CE_{pop} is the catch weighted CPUE in each of n_{pop} separate populations within each SAU (labelled as Blk), and CE_{Blk} is the average catch rate for a SAU. In a real fishery we do not yet have knowledge of each separate population (though this may change with the use of the GPS data loggers). Nevertheless, the current catch rates calculated for each SAU are certainly influenced by the number of records reporting landings. As there is a linear relationship between catch and effort then catch weighting the expected catch rates from each population provides an approximation to the usual method of calculating catch rates for the SAUs.

Once a weighted catch rate is available for each SAU in a zone, then each SAU then has a score calculated using:

$$S_{Blk} = trunc \left[0.1 \times CE_{Blk} - \left((CE_{targ} - 50) / 10 \right) \right] \quad (90)$$

where S_{Blk} is the score for each SAU Blk . The scores vary from 0 to 10 and the constants 0.1, 50, and 10 within this linear relationship lead to a score of 5 (the middle of the range) being attributed to the selected target catch rate (**Table 35**). The use of the function *trunc* converts each score into the lowest integer below the actual score. This ensures that the smallest change in TAC will be 2.5% up or down, whereas the maximum change can be 10% up or 20% down (**Table 34; Figure 96**).

Table 34. The TAC Multiplier for a given score. By modifying these responses it is simple to generate an alternative HCR. See **Figure 96** for a graphical representation.

Score	0	1	2	3	4	5	6	7	8	9	10
TAC Multiplier	0.8	0.85	0.9	0.925	0.95	1	1.05	1.075	1.1	1.1	1.1

The scores from each SAU (or block in Tasmania) are combined into a single zone score (S_Z) by weighting each separate SAU score by the proportion of the zone wide catch (the TAC) taken in each particular SAU:

$$S_Z = trunc \left[\sum_{Blk=1}^{nBlk} S_{Blk} \times C_{Blk} \right] \quad (91)$$

where C_{Blk} is the total catch in SAU Blk , and $nBlk$ is the number of SAU in a zone. The *trunc* function is again used to prevent the potential for very small TAC changes.

Table 35. How the catch rate scores vary for different observed catch rates when different target catch rates are selected. Note that the central score of 5 (highlighted cells) occurs when the observed catch rate equals the target catch rate. These scores derive from Equ (90).

Observed Catch Rates	Selected Target Catch Rates				
	60	70	80	90	100
20	1	0	0	0	0
30	2	1	0	0	0
40	3	2	1	0	0
50	4	3	2	1	0
60	5	4	3	2	1
70	6	5	4	3	2
80	7	6	5	4	3
90	8	7	6	5	4
100	9	8	7	6	5
110	10	9	8	7	6
120	10	10	9	8	7
130	10	10	10	9	8
140	10	10	10	10	9
150	10	10	10	10	10

The final zonal score is translated into a TAC multiplier using a simple look up table and the following year's TAC is set by multiplying the current TAC by the multiplier.

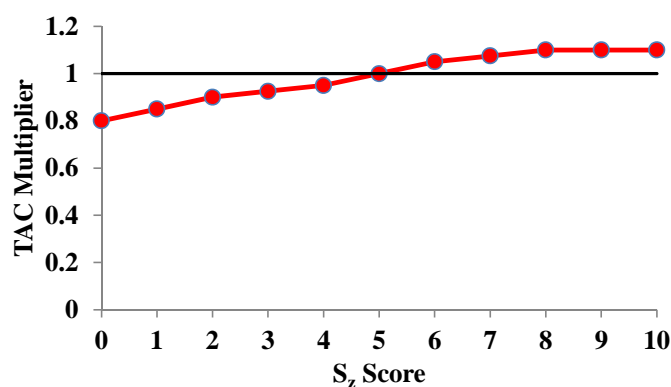


Figure 96. A graphical representation of the values translating a particular zone score into a TAC multiplier (see **Table 34**).

While this may appear an overly complex method for generating a TAC multiplier the translation into a score and the combination of multiple scores into one, sets up a framework for including as varied and as many different performance measures, each with their own score and weighting, as wished.

The catch rates used to determine the TAC in a year are those from the previous year, however, it would easily be possible to find some other method for gaining an estimate of current catch rates, perhaps something like the mean CPUE of the last three years or some other period, or some other method altogether.

21.2.4 A Lower Limit on the TAC

While in practice it might occasionally be reasonable and sensible to close a single block or SAU, even for a complete year, it is not expected that a requirement for a zero TAC would ever be required for a whole zone. However, a very low TAC might be recommended by a HCR but this might not be acceptable for reasons of maintaining processing infra-structure, markets, or other reason. In practice, in the Tasmanian abalone fishery, very low TACs have not been necessary throughout its history (although currently the eastern zone is at its lowest catch level ever). Despite this some of the HCR considered can, under some conditions, lead to recommendations for exceptionally low TACs. Rather than completely change the HCR, a simple solution is to add a decision rule that the TAC will not be allowed to fall below some minimum threshold. This is not the same as a limit biological reference point as it involves a restriction on a management action rather than reflecting some property of the fishery. Such a restriction is likely to greatly alter the behaviour of any HCR; such a restriction could be applied to almost any HCR, however, adding it in doubles the number of scenarios that we are likely to need to consider.

In the analyses here the selection of a lower TAC limit was implemented as a given proportion (50%) of the initial TAC used prior to the introduction of the HCR.

21.3 General Methods

21.3.1 Generation of a Simulated Zone

Simulated zones were generally made up of 70 separate populations and 10 statistical blocks (or spatial assessment unit – SAU) among which the 70 populations were distributed. Each block had seven populations but the productivity of each varied so blocks had widely different properties (see **Figure 97**). The observed data, and relationships between variables, reflecting biological variation from inside a selected fishing zone,

were used to condition each population to be similar to sampled populations. Details of the simulation are described in section 19.4.

The zone simulated was generally conditioned on the East coast of Tasmania, for which the most biological data is available. It was set up to have a biological minimum length (BML = size at 50% maturity + two year's growth) of approximately 138 mm; although this is the mean BML across the 70 populations. In addition it had a theoretical productivity, as measured by the maximum sustainable yield of about 632 t. Both these properties influence what scenarios should provide the greatest contrast in behaviour between HCR (if there are any differences).

21.3.2 General MSE Methods

A standard approach when using management strategy evaluation to compare and test alternative management strategies is to compare their relative performance at achieving the fishery objectives. With abalone the objectives remain relatively informal so such usual comparisons are difficult. Nevertheless, there are a number of options open to be pursued. Various management strategies using catch rates have been proposed as mimicking the informal processes undertaken in the discussions that surround the production of management advice in Tasmania and Victoria. Whether these can perform as intended (at least a sustainable fishery with management of catch to prevent depletion) can be determined in various ways. There are three potential steps: 1) deterministic zone simulations, 2) Single zone simulations, and 3) replicate zone simulations.

21.3.3 Deterministic Zone Simulations

One option is to run the MSE simulation framework in the absence of stochastic variation to determine the average behaviour of the strategy (see **Figure 98**). This provides for a rapid overview of the management strategy's performance over a range of conditions, which allows the identification of plausible candidates for future management without using further time on those management strategies (as defined formally in the MSE) that clearly fail to meet their intentions with their average behaviour.

SigmaCE, sigmaB, and sigmaR are the standard deviations of the distributions around the values of CPUE and exploitable biomass where those are used in considering performance and in distributing catch among blocks and populations, and the standard deviation of the log-normal distribution of recruitment variation. In the deterministic simulation runs these are set to a very small number ($1e^{-6}$), which has the effect of appearing to introduce no variation in the values of the variables in which they are used. All other aspects of the simulations remain unchanged. Only one run is required of the simulation as the behaviour of the model will be the same whenever it is run. Such simulations have value as, by removing variation, they identify the average behavioural response of the management strategy under whatever conditions the simulation is run under.

21.3.4 Single Zone Simulations

The only difference between these and the deterministic simulations is that the variation terms are all given positive values (see **Figure 99**). This, obviously, allows insight into the behaviour of the management strategy in the face of random variation in the dynamics. By providing particular estimates of the variation likely to be experienced in any single run the possibility of the averaging behaviour of using many replicates (see below) hiding properties (such as oscillatory dynamics) can be considered. This can also be pursued by only running a small number of replicates (such as only five).

One potential problem with the usual summary of MSE simulations is that it is often the asymptotic behaviour which is illustrated (see the next section) but this can obscure any transient behaviour at the introduction of a new management strategy and obscure details hidden by only viewing the variation deriving from numerous replicate zones.

21.3.5 Replicate Zone Simulations

The use of 100 replicate zone simulations to represent the full range of behaviours from each management strategy reflects the standard usage within MSE testing, where the asymptotic average behaviour and the range of surrounding variation is of primary interest (see **Figure 100**). This allows alternative management strategies to be compared in a probabilistic sense by characterizing the proportion of runs that achieve an objective. These will only be applied to those management strategies that meet the minimum performance required to satisfy inspection in the single zone simulations (both with and without variation).

21.3.6 Scenarios Considered

Six different HCR were considered including the CPUE gradient, the CPUE gradient plus the TAC perturbation, and the target CPUE HCR. Each of these three HCR was then modified by using a TAC lower limit. By combining three states of initial depletion, three LML, and three initial TAC this implied that 27 separate scenarios were examined for each HCR giving a total of 162 scenarios (**Table 36**).

Table 36. The initial 162 scenarios considered, including the six different HCR and the 27 different combinations of initial depletion, LML, and initial TAC that were used to bracket the possible conditions under which the HCRs might be applied.

	TAC Limit	No TAC Limit
CPUE Gradient	27	27
+TAC Perturb	27	27
Target CPUE	27	27
Initial Depletion	LML	Initial TAC
Low	127	Low (450 t)
~Production	132	~Production (600 t)
High	138	High (800 t)

When the CPUE Gradient HCR was used the default period for calculating the gradient of catch rates was five years. With the Target CPUE HCR the target was set at 85 kg/hr across the zone. This catch rate was chosen as representing two periods since 1997 generally recognized as having the east coast resource in good condition.

Once these scenarios had been used to determine the types of behaviour exhibited by the different HCR alternative periods and targets were examined to determine whether such alternatives interacted with the HCR and the abalone dynamics to alter the behaviour exhibited with the default period and target.

21.3.7 Depletion of a Zone to High, Target, and Low Status

Before any MSE were run the simulated zones were each depleted to some given level. These levels were all relative to the level at which the simulated zone was expected to be the most productive. An outcome of the conditioning (the 70 populations given properties similar to east coast Tasmania) is that the simulated zone should be most produc-

tive at an overall depletion of its spawning or mature biomass at a level of about $37\%B_0$. Examples were considered where the zone had been depleted to three states: a relatively low level of spawning biomass (relative to that which would produce the maximum sustainable yield; $\sim 30\%B_0$), a middle level of depletion which would be only slightly above maximally productivity ($\sim 40\%B_0$), and a relatively high level of spawning biomass, above the MSY levels ($\sim 50\%B_0$). The exact levels varied between 1 – 3% because of how the simulated zone was depleted and the effect of fishing a zone at three different LMLs (**Table 37**).

It was found that there is a roughly linear relationship between the states of depletion obtained from a zone fished at a particular LML with a constant harvest rate imposed for 50 years. A Beverton-Holt stock recruitment relationship is used to calculate each year's recruitment and for this fish down period this was used with no stochastic variation. By trial and error it was possible to calibrate these relationships, calculate a linear relationship between the harvest rate of exploitable biomass and the desired depletion level after 50 years fishing and use that to obtain approximately the specified depletion level for ranges between 25% and 80% unfished biomass. The exploitable biomass was always less than the spawning biomass.

Table 37. The depletion levels achieved for both spawning and exploitable biomass prior to exploitation under control of each HCR. The columns relate to three alternative LMLs. The values under the spawning and exploitable columns are the zone-wide depletion levels relative to unfished levels. The top three rows relate to depletion to a low level of spawning biomass, the middle three to about the level of maximum productivity, and the bottom three to a relatively high level of spawning biomass.

Spawning	127	132	138	Exploitable	127	132	138
0.271			X	0.171			X
0.299		X		0.205		X	
0.302	X			0.207	X		
0.373			X	0.287			X
0.402		X		0.329		X	
0.404	X			0.330	X		
0.494		X		0.430			X
0.497			X	0.436		X	
0.503	X			0.445	X		

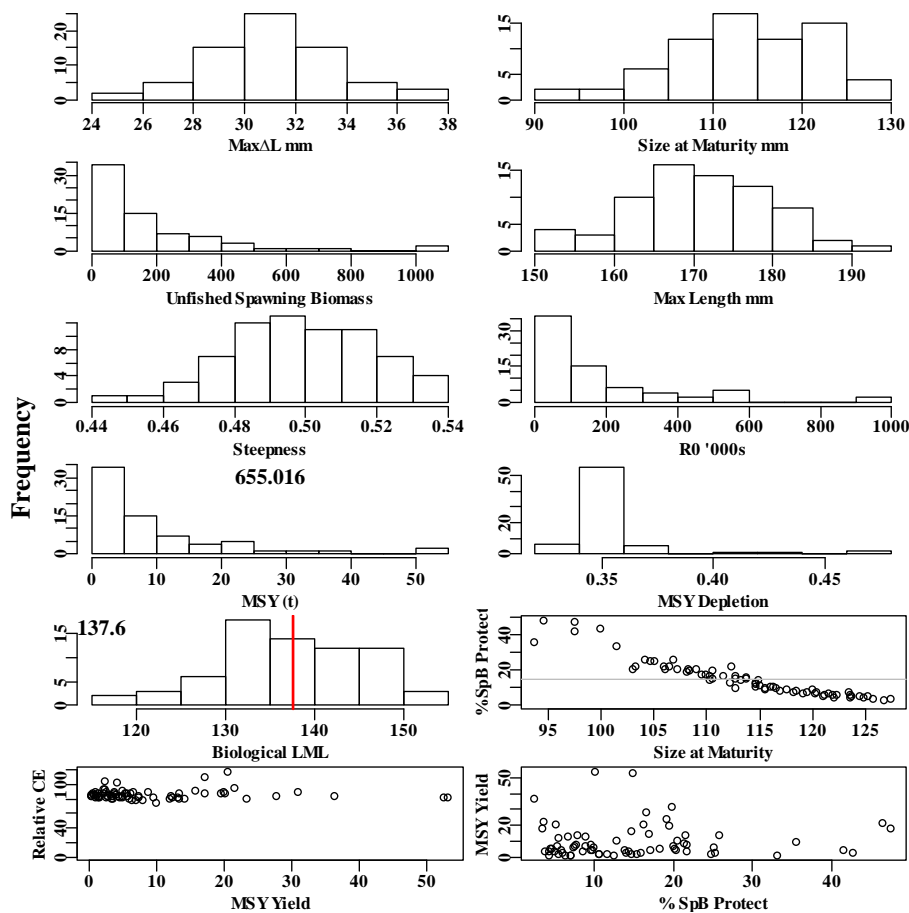


Figure 97. Some of the properties of the 70 populations, in their unfished state, that make up the simulated zone used in the following simulations. The Max Δ L is one of the growth curve parameters for each population. The steepness and R0 are both parameters relating to the Beverton-Holt stock recruitment curve. The MSY depletion is the population depletion level where the population is at its most productive.

There was no requirement to try to deplete each scenario to exactly the same level because following the initial depletion the simulated zone was fished for 10 years at a constant initial TAC but with recruitment variation turned on to its pre-set level, with the objective of introducing a full range of variation to the stock before any comparisons would be made. In addition, so that the TAC could be allocated as it would normally be in the simulated fishing, the variation associated with catch rate observations and exploitable biomasses were also turned on. The effect of doing this was that the zone took on a more realistic variation of state between populations and SAU combinations of populations prior to the HCR under exploration being turned on (e.g. **Figure 99**).

21.3.8 Alternative Initial TAC

The theoretical MSY of all populations in the conditioned zone combined was about 665 t. However, when these separate populations are combined into SAU it is not always possible to fish each population to its most productive state because of differences between component populations within a SAU. Because of this the maximum that appears possible from a whole zone appears to be about 630 t if the zone is managed using fishery statistics from the SAU rather than individual populations. Initial TACs of 450, 600, and 800 t were therefore used to impose rather different conditions on the zone, which would provide for a broad range of conditions over which to test the effectiveness of the different HCR.

21.3.9 The Simulated Zone

The conditioning of the simulated zones produces a plausible range of parameters across the component populations (**Figure 97**). The initial unfished simulated zone is identical in all scenarios except in relation to its productivity. The LML selected for a particular scenario will have an influence on the proportion of the spawning biomass protected by the LML. It will also influence the selectivity of fishing, which will in turn have an influence on the maximum sustainable yield (MSY) that the zone can produce, the state of depletion that would produce the MSY, and hence the catch rates that are expected to be produced when fishing at the MSY (**Table 38**). Most populations only have relatively minor yields but the properties of each of the 10 blocks, which combine seven populations, are less variable than the range of individual populations (**Figure 98**; **Table 39**).

Table 38. The differences between the simulated zones depending on the average biological minimum length (BML) and the LML imposed when fishing. The % B_0 Protect is the percent of the unfished spawning biomass protected by the LML. msyDepletion is the theoretical percent of the unfished spawning stock for it to produce the MSY and msyCPUE is the expected catch rate at the MSY.

BML	LML	% B_0 Protect	MSY (t)	msyDepletion	msyCPUE
138	127	8.834	627.911	36.66	76.61
138	132	14.864	633.338	36.91	72.30
138	138	23.217	632.795	37.98	65.55

To provide an example of the outputs from such a simulated zone (illustrated in **Figure 97**; **Table 39**), a zone was fished to an initial state of depletion of approximately $0.3B_0$ and then fished for ten years at a TAC of 600 t at an LML of 127mm, after which the Target CPUE HCR was introduced with a target of 85kg/hr and the zone fished for another 40 years (**Figure 98**). Stochasticity was omitted to illustrate the underlying dynamics.

The spawning biomass averaged across all 70 populations was $33.3\%B_0$ in year 10 of the initiation (with a standard deviation of 0.224), but across the last 20 years (30 – 50) it reached $\sim 47\%B_0$. The variation apparent across populations (in spawning biomass and catch rates) reflects the range of properties given to each population when the zone was generated. Even though the stock size and associated catch rates begin well below the target of 85 kg/hr (indicated by the horizontal black line in the two CPUE graphs) the HCR operates to alter the TAC and drive the CPUE up towards the target. There is a time lag between altering the TAC, which changes the spawning biomass, and the growth of new recruits into the fishery, which will change the CPUE. This time lag leads to under- and over-compensation so the CPUE over- and under-shoots the target and proceeds to approach it via declining oscillations through time. After the introduction of the HCR it takes the fishery about 10 years for the zone average CPUE to first reach the target CPUE; the average over the final 20 years was 86.7 kg/hr. The TAC declines rapidly initially, but doesn't quite reach the limit of 50% of the original TAC but stays stable at 310 t in years 18 – 21. The effect of the scoring by integers within the HCR is apparent in the step like proportional changes that the TAC undergoes through the 40 years of the operation of the HCR. Finally, without recruitment variation, the predicted recruitment is directly related to the spawning biomass (**Figure 98**).

Table 39. Individual block properties within a simulated zone. SaM is size at 50% maturity, AgeM is the implied age at maturity, B0 is the unfished spawning biomass, ExB is the unfished exploitable biomass, MaxL is the size of a 23 year old abalone, BML is the biological minimum length (SaM + two year's growth), MSY is the theoretical maximum sustainable yield, MSYdepl is the depletion level of the spawning biomass when it is capable of producing the MSY, MSYCE is the expected mean catch rates when the fishery is producing at B_{MSY} , and ExpCE is the expected unfished catch rates for each block. The bottom row is either the average (columns 2, 3, 6, 7, 9, 10, and 11) or the sum (columns 4, 5, 8) of the columns for each block.

Block	SaM	AgeM	B0	ExB	MaxL	BML	MSY	MSYDepl	MSYCE	ExpCE
1	115.539	4.795	2728.407	2385.411	190.804	139.976	137.756	0.346	81.681	235.961
2	107.617	4.590	902.843	702.980	184.475	132.208	45.626	0.352	82.957	235.988
3	114.285	4.782	1160.579	1006.892	182.135	138.405	57.815	0.353	81.881	231.859
4	110.706	4.997	1169.454	873.555	176.234	133.140	60.230	0.371	88.929	240.990
5	117.415	4.725	1029.135	919.703	182.789	142.640	50.359	0.35	83.943	240.087
6	110.971	4.824	1166.391	958.132	178.590	134.678	60.151	0.356	84.146	236.916
7	118.547	5.145	451.307	395.396	185.643	141.840	23.782	0.341	83.166	244.282
8	117.502	5.018	2097.641	1892.368	179.497	141.036	102.778	0.351	86.599	247.205
9	114.525	4.662	1433.654	1235.991	186.974	139.888	69.095	0.354	85.030	240.440
10	109.159	4.843	936.359	681.794	176.224	132.091	47.423	0.379	90.670	241.382
Zone	113.627	4.838	13075.77	11052.22	182.337	137.590	655.015	0.355	84.900	239.511

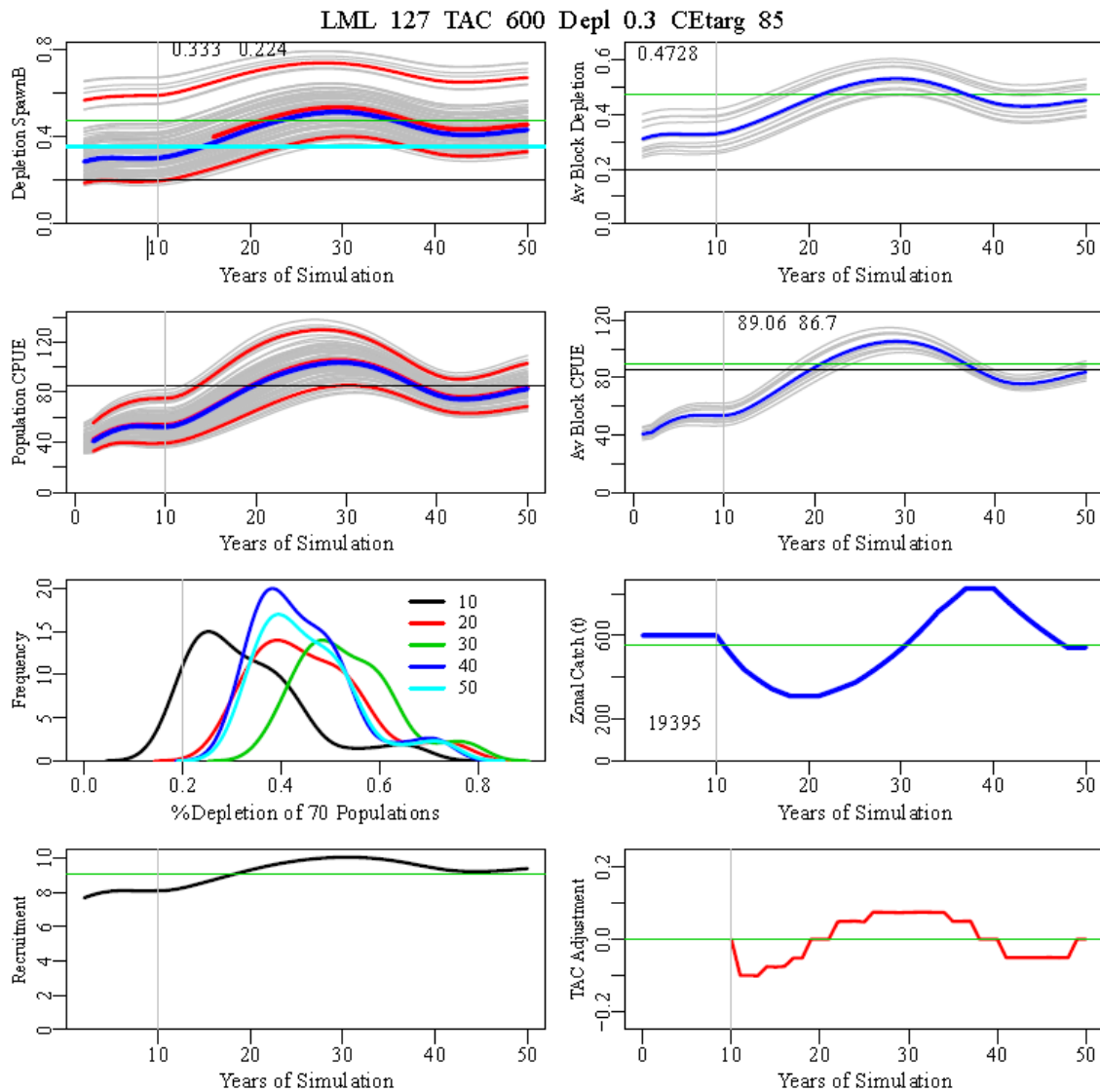


Figure 98. A single simulated zone initially depleted to $\sim 0.3B_0$, fished at 600t at an LML of 138mm and target CPUE of 85kg/hr. Graphs from top left to bottom right are: 1) population depletion levels; 2) block depletion levels; 3) population catch rates; 4) mean block catch rates; 5) depletion distributions in years 10 – 50; 6) annual catch (t), with total catch from years 16 – 50; 7) the zone wide recruitment series; and 8) the TAC Adjustment. The population graphs include the median and 90% quantiles. The block graphs include the mean of years 16 – 50 as green lines, with the average block CPUE also showing the mean from years 30 – 50.

When stochastic noise is included into the dynamics of the fishery (estimates of exploitable biomass and CPUE), and recruitment variability is included then the general trends and patterns remain similar to those in the previous graph but now the overall catch across the last 35 years (years 16 – 50) is $\sim 3,000t$ less than the deterministic modelling. In addition, at about year 36 there is a zone wide recruitment event, which aids rebuilding between 42 – 46 years (this may not occur, or may occur at a different time, in a different run). Also the TAC adjustments exhibit a much less predictable behaviour, so that the integer nature of the scoring is now very much less apparent. In terms of the HCR performance the management appears to be less stable so that achieving and maintaining the target is more difficult; although in this particular case the CPUE falls above the target more often than below in the last 20 years (hence the average of 90.6 kg/hr).

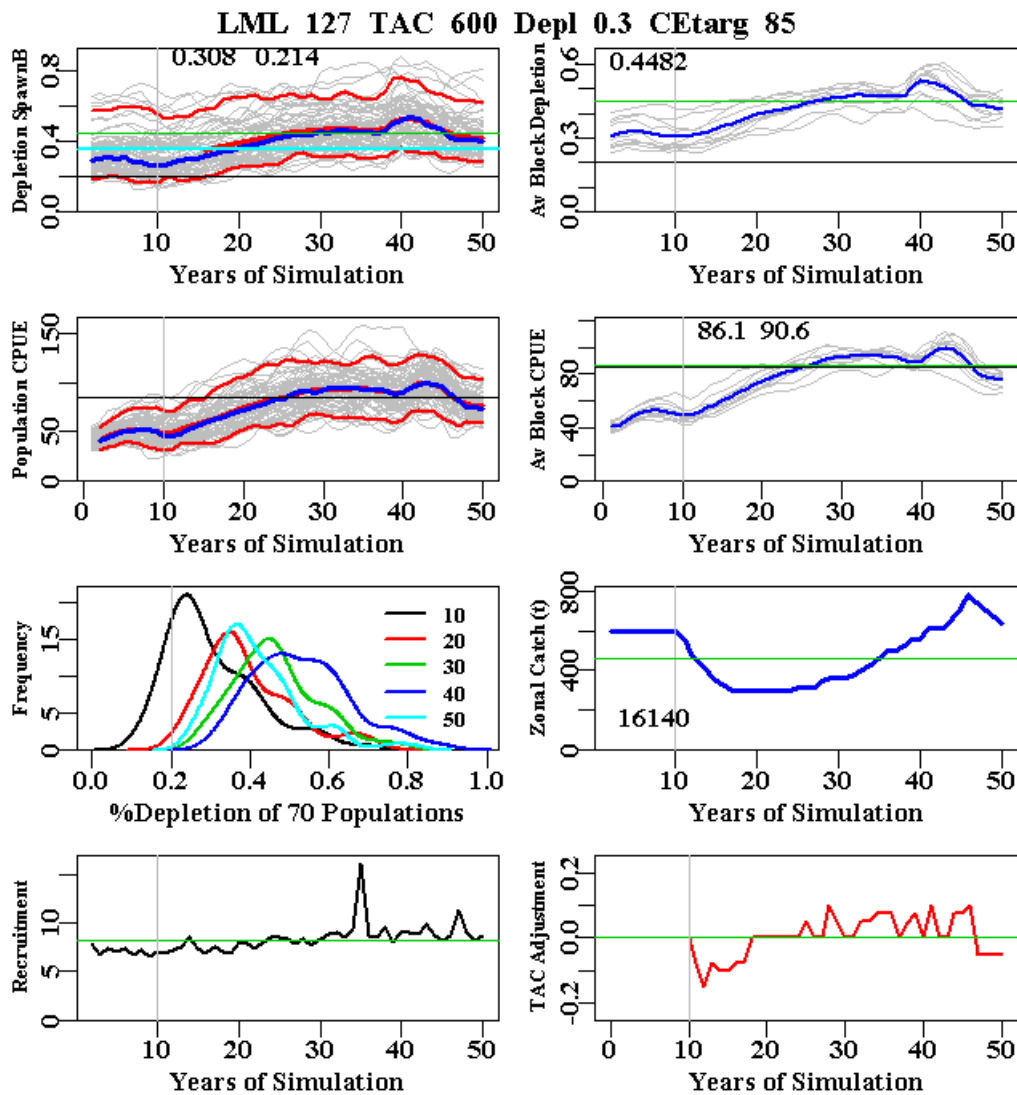


Figure 99. The same zone as used previously (**Figure 98**) except that variation is included with the estimation of exploitable biomass, fishery data on CPUE, and on recruitment, hence the non-smooth lines.

21.3.10 The Geographical Scale of Application of the HCR

Current fisheries data is not available at the population level and in both the real world and the simulations the fisheries data and consequent management operates at the scale of SAU or statistical blocks. It is immediately obvious that the variation in spawning biomass depletion level and catch rates is much greater when considering individual populations than when considering SAUs (consider the different y-axes in the top four graphs in **Figure 99**). Variation is introduced from three main sources which include variation in the observed catch rates from the divers, variation in the distribution of catches relative to the available biomass in each block, and recruitment variation around the Beverton-Holt recruitment relationship. Some of the changes in the dynamics visible through time are clearly a time-lagged response to a peak or drop in recruitment.

21.3.11 Replicate Zones

Single replicate simulated zones, as in the last section, can vary greatly between replicates of exactly the same scenario due to the random variation influencing the particular sequence of recruitments and fishing events. While replicate scenarios are likely to be similar they would only very rarely generate the same outcomes. To understand the

overall expected behaviour of the various HCR versions it is therefore necessary to run many replicates of each simulated scenario. Each simulation generates 70 populations and 10 blocks, so 100 replicates would produce the dynamics of 7000 individual populations and 1000 separate blocks, which is sufficient for the purpose of characterizing differences between scenarios and between HCR; for this reason the default number of replicates was set at 100.

An example, where a simulated zone with a BML of about 138 mm was run 100 times with an average initial depletion of about $0.3B_0$, and initial TAC of 800 t, and a LML of 127 mm, provides an illustration of the type of outputs possible (**Figure 100**). Once again the variation declines from population to block, but also when summarizing the 100 replicate zones.

As a result of the variation in the size at maturity affecting the proportion of the spawning biomass protected by the LML, while the mean depletion level is about $0.25B_0$ some of the underlying populations and blocks have rather different spawning biomass depletion levels (**Figure 101**). This variation is not seen in the depletion levels of the exploitable biomass because this only relates to the available biomass above the LML, hence the initial catch rates also show a more coherent set of values (**Figure 100**).

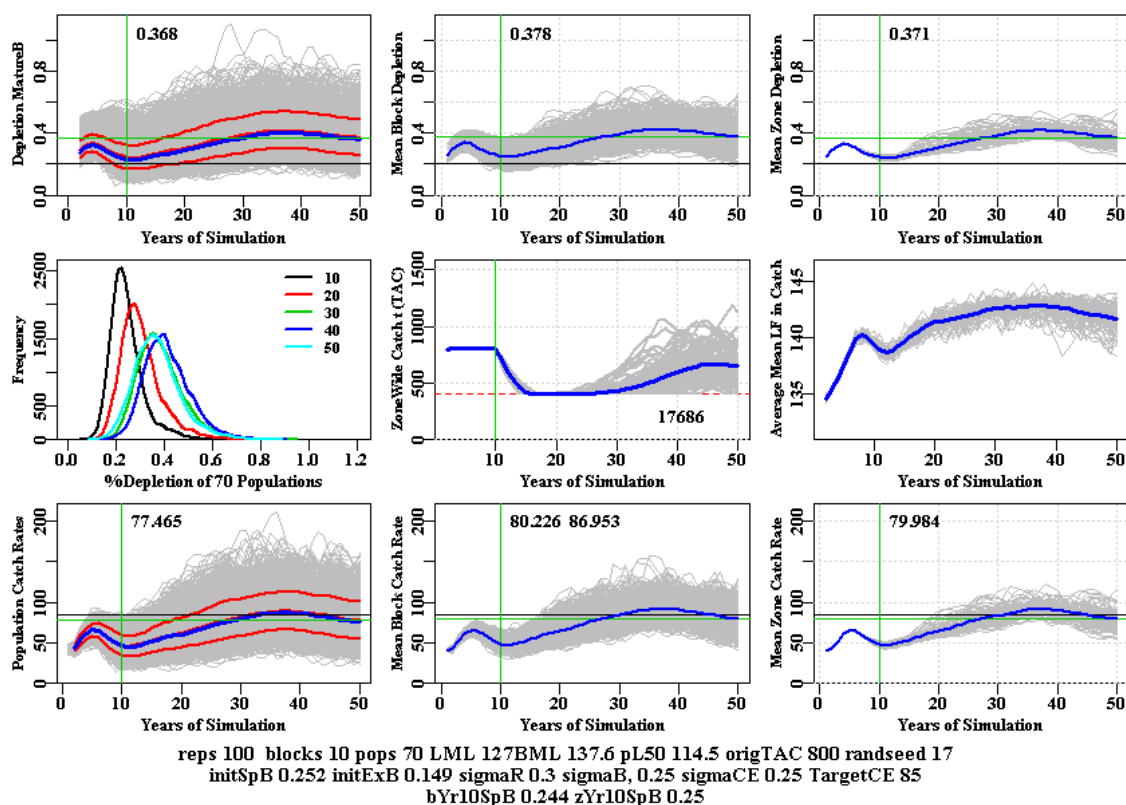


Figure 100. The outcome from 100 replicates of a zone with a BML of 138mm, fished at 127mm at an initial depletion level of $0.25B_0$ and an initial TAC of 800 t. The numbers in each case refer to the mean across years 16 – 50 or, in the catch graph, to the total average catch from 16 – 50 years. The horizontal green lines visible in the bottom three graphs is the Target CPUE. The conditions for the simulation run are listed in the text at the base of the figure.

With the initial conditions of this example run, in 91/100 replicate zones catches rapidly decline down to the limit TAC of 400 t (0.5×800 t) and they do that generally within 5 – 6 years of introducing the HCR (though some take only 4 years and some take 10; **Figure 100** and **Figure 102**). The period over which they remain at the limit is much

more variable and ranges from 1 to 17 days with a variety of dominant periods. In some instances it appears the catches recover only to decline back down to the limit, but in fact the data indicates that they are just above the limit.

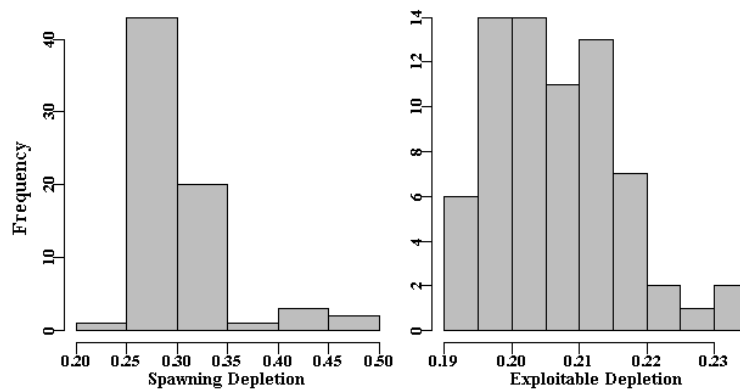


Figure 101. The distribution of depletion levels for both the spawning biomass and the exploitable biomass across 70 populations when the average depletion level for the spawning biomass was $0.3B_0$.

Importantly, the average catch-rates over years 16 – 50 (the final 35 years) is about 15kg/hr above the target used in the target CPUE HCR. Once again it appears that time lags involved in using catch rates to recommend TACs lead to delays in the dynamics so that targets can be over-shot and take a while for compensation to come into operation.

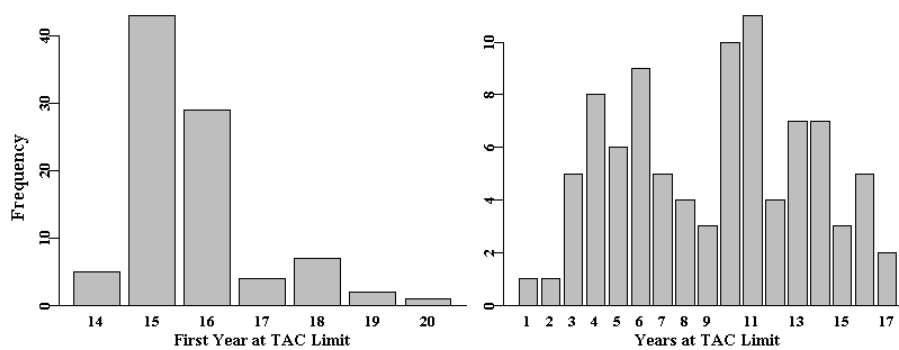


Figure 102. For the example scenario described in **Figure 100**, this is the distribution of the first year in which the HCR takes the TAC down to the TAC limit of 360 t and then the distribution of the duration of staying at that lower limit.

21.3.12 Diagnostic Statistics

The diagnostic statistics used in Management Strategy Evaluation studies for distinguishing between alternative HCR are often calculated as the distribution of replicate outputs relative to some target or limit reference point or relative to some quantity of interest generated by the MSE simulations (for example, see Punt *et al.*, 2002). This is possible with the CPUE Target HCR but not with the two others. Nevertheless, there are a number of diagnostic statistics

(Table 40) that can be used to provide detailed comparisons between the HCR being compared under the different scenarios of initial conditions.

While it is usually true that each HCR newly applied to a fishery will eventually generate asymptotically stable behaviour an important aspect of MSE simulations that is often neglected is the transient behaviour that can be expected to occur on the new HCR's introduction. With the abalone simulations the results are more than simply variable as through time there can be long-term oscillations (e.g. Figure 100), which appear to be related to the time lags that exist between recruitment as post-larval forms and entering the fishery between 5 – 7 years later. One way of attempting to capture the presence of such transient behaviour is to calculate diagnostic statistics for both the first 10 years following the model initiation (years 11 – 20), and for the final 30 years of the simulations (from years 21 – 50). The first period will include any immediate changes brought about by the HCR introduction and the second period will include any longer term behaviour of the fishery when managed with the given HCR.

Given the two periods it is possible to consider the mean and variation of TACs or catches, the catch rates, and the state of spawning biomass depletion. In case the frequency distribution of such model outputs differs greatly from a normal distribution (so an arithmetic mean may not represent the central tendency very well) we can also generate the median estimate and the inter-quartile distance (the gap between the 25th and 75th quantiles)(Table 40).

Table 40. Diagnostic statistics used to compare alternative scenarios of conditions and HCR. InterQ is the interquartile distance, the Freq is the frequency of replicates in which a given threshold is breached, and the Duration is the number of years over which the threshold breach occurs. The depletion is of spawning biomass, and %ΔCPUE is the percent change in TAC during the period following the introduction of the HCR. Where there is a target for CPUE the performance relative to that target will also be characterized as a diagnostic.

Diagnostic	Catch/TAC		CPUE		Depletion		%ΔCPUE	
	21-50	11 - 20	21-50	11 - 20	21-50	11 - 20	21-50	11 - 20
Mean	X	X	X	X	X	X		X
Variation	X	X	X	X	X	X		X
Median	X	X	X	X	X	X		X
InterQ	X	X	X	X	X	X		X
Freq < Const	X	X	o	o	o	o		X
Duration < Const	X	X	o	o	o	o		
Freq > Const	o	o	X	X	o	o		X
Duration > Const	o	o	X	X	o	o		
Relative to Target			X	X				
Years	21-50	11 - 20	21-50	11 - 20	21-50	11 - 20	21-50	11 - 20

In addition to such statistical summaries of simulation outputs it is also possible to quantify the ability of the various HCR to avoid certain poor management outcomes or become overly optimistic. For example, the scenarios include the option of having a lower limit on the TAC can be formally characterized through determining in how many replicates this limit is reached or breached and in what year of the simulation this first happens. As well as this frequency and first occurrence, the duration of breaching the

limit within each replicate can also be tabulated. These data, the year of occurrence and the durations will produce distributions from each HCR, which can then be compared in terms of their range and medians to determine which HCR can produce the best management outcomes. If thresholds are also set for an upper limit on TAC and lower and upper limits on CPUE and spawning biomass depletion then these two measures can be characterized from count, year of occurrence, and duration (**Table 40**).

The TAC is the main management lever being explored in this simulation tests. When considering the transients associated with introducing a new HCR, as well as characterizing the diagnostic statistics for the early period in the simulation it is also possible to tabulate the rate of change and variability of the TAC, as well as whether it breaches some chosen limit, during the first 10 years following the introduction of the HCR.

21.4 Results

21.4.1 Gradient CPUE HCR

There is no explicit fishery objective associated with the HCR that operate with respect to the gradient of catch rates, although there is an implicit target of a zero CPUE gradient. The HCR simply reacts to how catch rates change and are used as performance measures in the informal assessments currently underway and also in the multi-criterion decision analysis being proposed in Tasmania. In stock assessment situations there are some indications that similar CPUE gradient HCR can operate with some success (Dichmont and Brown, 2010). In this case, successful operation would constitute the HCR moving the fishery away from a depleted, low catch rate state to a less depleted higher catch rate state. Without a specified target boxplots of the final outputs relative to expected performance are not appropriate to illustrate the behaviour of the HCR, instead, using average trajectory plots of the 100 replicates for each zone will be used (e.g. **Figure 103**).

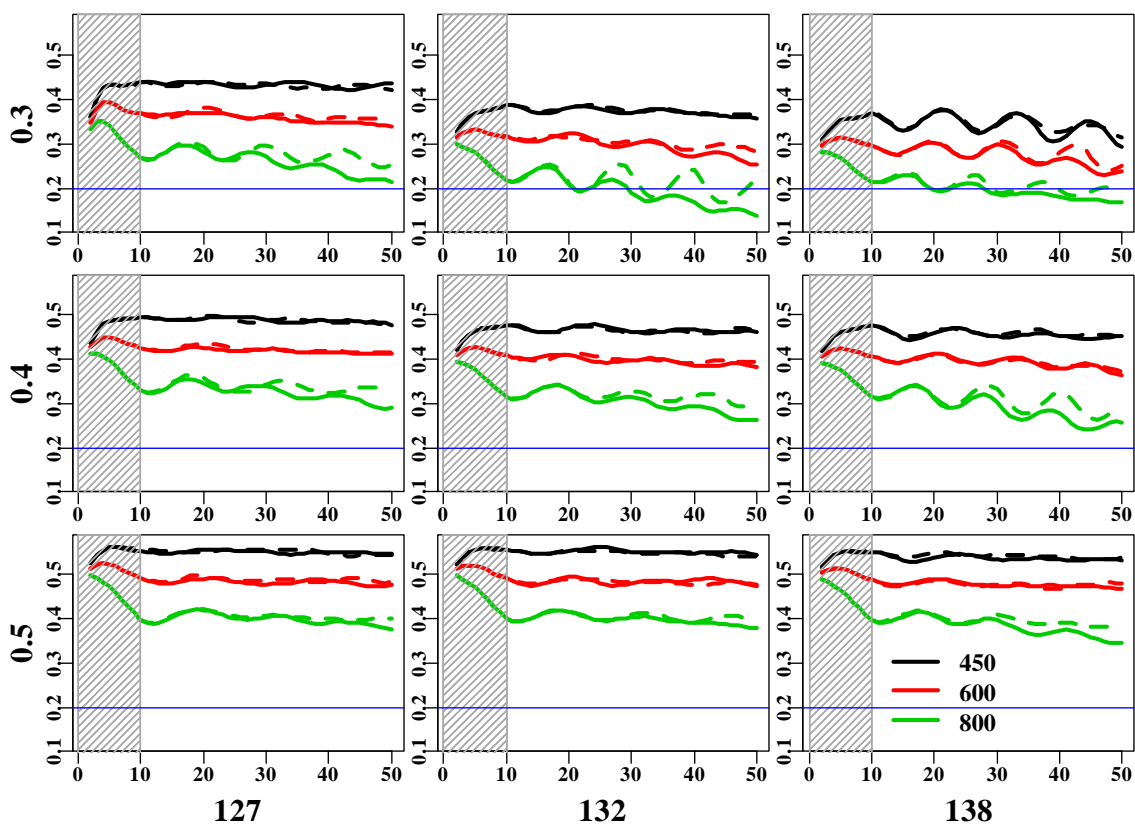


Figure 103. Spawning biomass depletion levels under 54 different scenarios considered with the Gradient CPUE HCR (3 initial depletion levels – y-axis, the three LML – x-axis, and the three initial TACs). The dashed lines relate to where no lower TAC limit is imposed and the grey box is the period of zone initiation.

Unfortunately, unlike crustacean fisheries and some finfish fisheries, with abalone the use of the CPUE gradient HCR generally fails to operate in any useful way. While it can maintain stability under almost half the initial conditions, the status quo is only useful when the situation happens to equate to catches being about the same as the maximum yield. In almost all cases a TAC of about 75% of the MSY, 450 t, or almost at the MSY at 600 t, the outcome was stability at whatever level of spawning biomass the zone was at when the HCR was introduced (**Figure 103**). Only with a LML of 138mm and initial depletion of 30% were oscillations induced as part of a slow decline in abundance. Although slight oscillations in abundance and possibly a slow decline was apparent with

combinations of LML 132mm and an initial depletion of 30% or a LML of 138mm (**Figure 103**). With a TAC of 800 t, declining abundance and oscillations in abundance were apparent with initial depletions of both 30% and 40%, with the oscillations and declines being more severe at the 30% initial depletion. Where there was a no TAC limit imposed the final reduction in abundance was not so great as when there was a limit. Oddly, the worst case for oscillations in abundance and final depletion level was a combination of 30% depletion and a LML of 132mm with a TAC limit imposed (**Figure 103**). This appeared to be due to the degree of over- and under-compensation being greatest with this combination.

In terms of spawning biomass the CPUE gradient HCR can lead to stability where catches are well below maximum. However, in the more realistic situation where catches are being maximized, at best, this HCR may slow depletion so that while it does occur it only does so relatively slowly. A TAC Limit has the effect of increasing the rate of decline.

The predicted catch rates for the 54 scenarios follow very similar trajectories to those of the spawning biomass, which implies that, in this case, the exploitable biomass is closely correlated to spawning biomass in all cases (**Figure 104**).

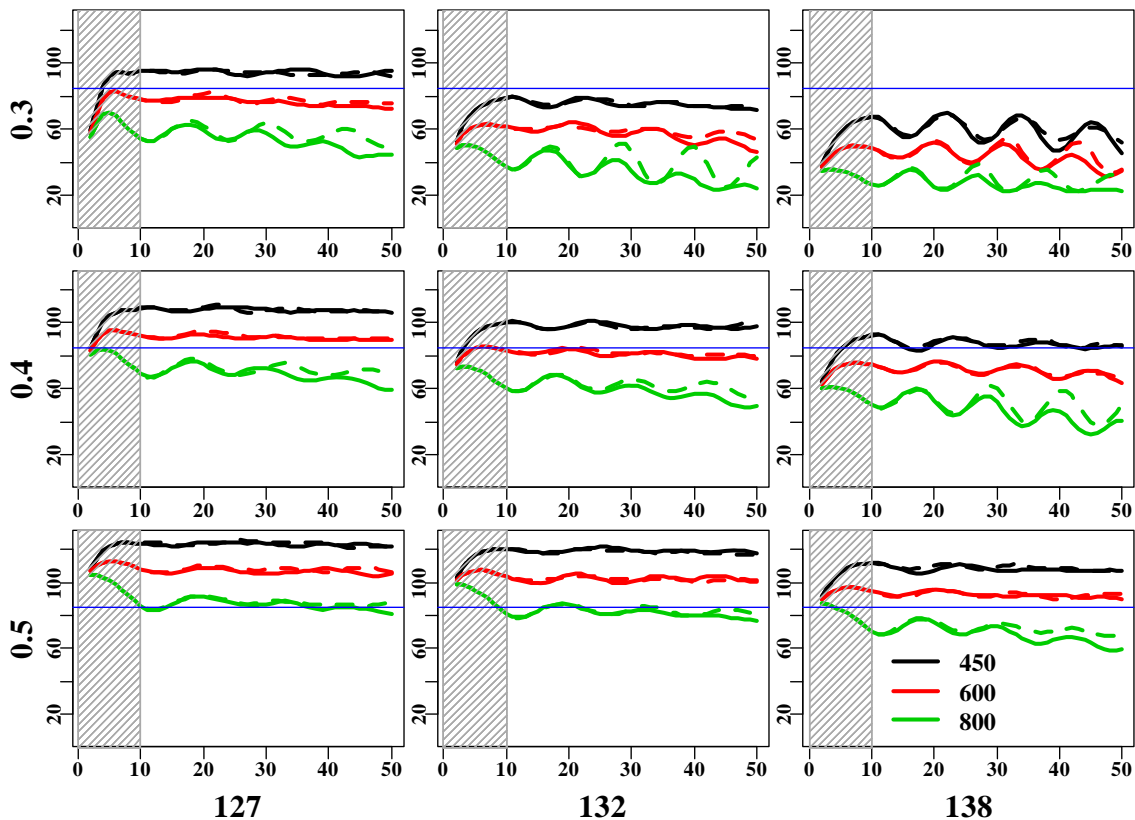


Figure 104. CPUE performance under each scenario using the CPUE Gradient HCR. The dashed lines relate to where no lower limit is imposed on the TAC.

In terms of catch rates, the use of the CPUE gradient HCR appears to lead to oscillations that have a remarkable correspondence with those observed recently on Tasmania's east coast. The CPUE gradient HCR is largely a status quo strategy, except in situations of greater degrees of depletion or higher LML. If the CPUE gradient HCR is used to produce management advice that is followed (presumably even if used informally), then the larger the LML the less stable the management outcome (**Figure 104**).

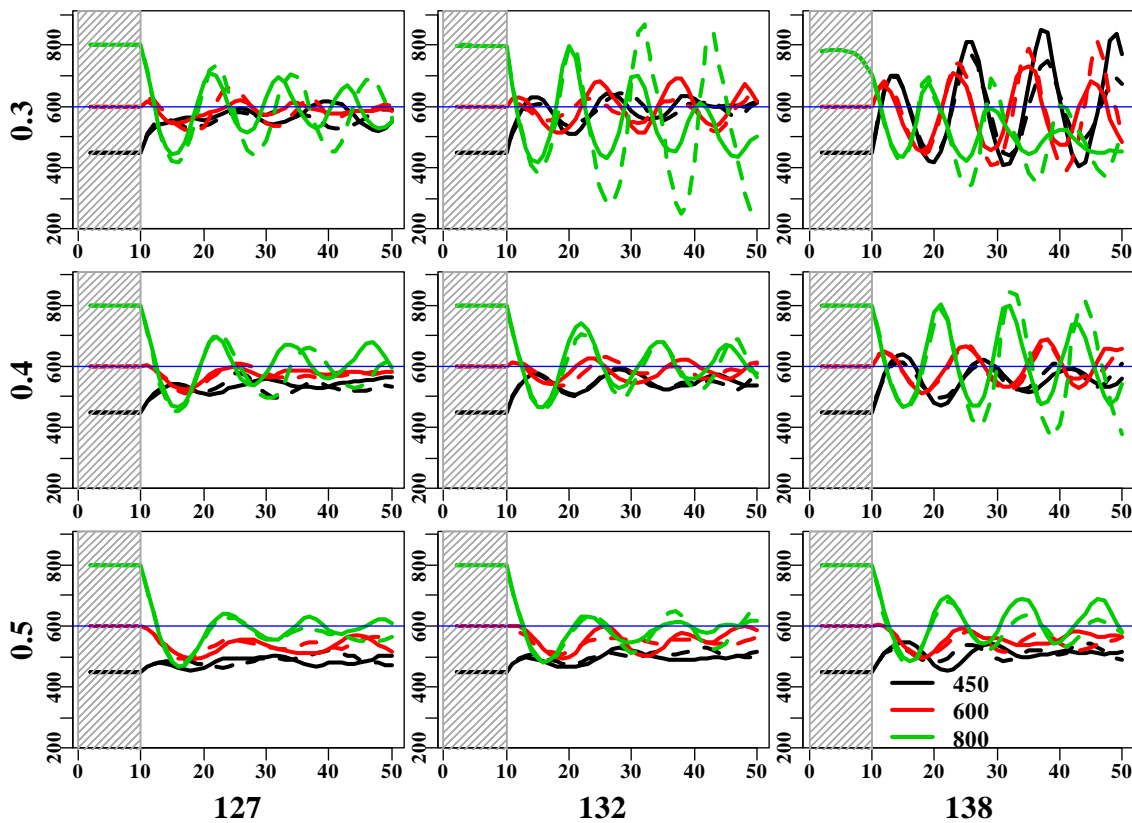


Figure 105. Total catches taken in the 54 different scenarios under the Gradient CPUE HCR. The dashed lines relate to where no lower limit is imposed on the TAC. The blue line is at 600 t, just under the theoretical MSY for the zone

Zone catches under all scenarios exhibit oscillations with the most extreme cases being in the 132mm and 138mm LML with the 30% and 40% initial depletion combinations, especially those where there is no TAC limit imposed (**Figure 105**). Even the initial TAC of 450 t can lead to large oscillations brought about by severe over- and under-compensation brought on, in turn, by time lags occurring between changes in TAC and response in the CPUE. The time lags are simply due to the time it takes for any reductions or increases in recruitment brought about by decreases or increases in spawning biomass caused by increased or decreased catches. The changes in catch brought on by the introduction of the HCR, not surprisingly, are least with the initial TAC of 600 t, which is closest to the MSY.

The most severe situation is the initial depletion of 30%, initial TAC of 800 t at a LML of 138mm. Catches of 800 t for initial period of 10 years cannot be maintained as the available exploitable biomass drops to too low a level. If the CPUE gradient HCR was to be used for management it appears to suggest that a LML of 127 mm would be the safest option. This does not match experience with the Tasmanian fishery, which undoubtedly became highly depleted in the 1980s at a LML of 127mm state-wide. However, the simulations here were limited to an initial depletion of 30% and a maximum initial TAC of 800 t, which is about 27% greater than the theoretical MSY. Further work on the predicted dynamics at higher catch levels might indicate the possible scenarios experienced by the fishery at that time.

In most combinations the contrast in the dynamics is greatest with the highest initial TAC. The initiation period of 10 years at this TAC always depleted the initial depletion level further, so this leads to the most stressful situations. Comparing the 9 scenario

combinations that include an initial TAC of 800 t highlights the differences between the alternative initial depletion levels at different LML (**Figure 106**).

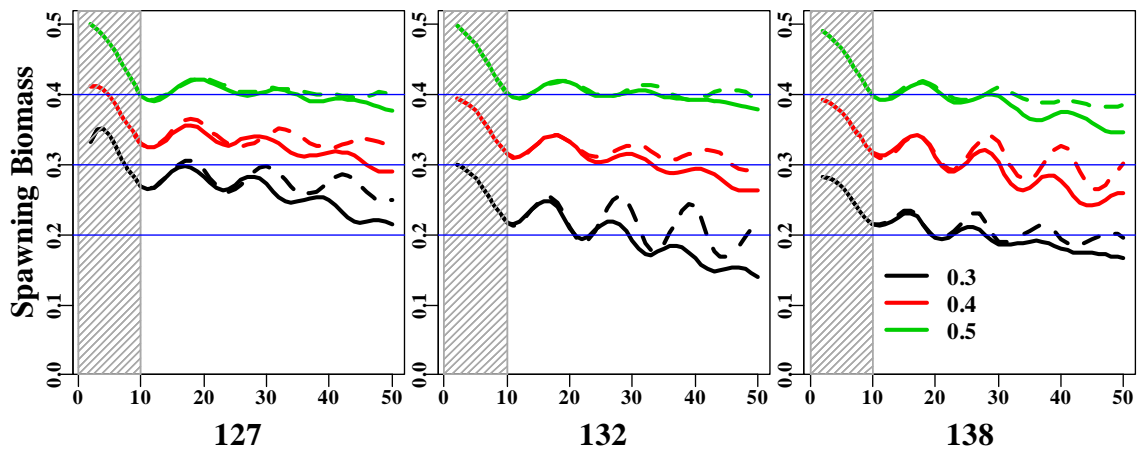


Figure 106. A direct comparison of the effects on the spawning biomass depletion levels of different initial depletion levels (see legend) when the three different LML have an initial TAC of 800 t. The three fine blue lines are to aid comparisons.

In all cases with an initial TAC of 800 t, the spawning biomass declines slowly through time with the rate of decline generally being greater the more the zone was depleted initially. The rate and extent of decline is also greater the larger the LML, although, only for the initial depletion level of 30%, the most severe final depletion and greatest variability in spawning biomass was in the 132mm LML combination. The positive influence on the spawning biomass depletion levels of not having a TAC limit is more apparent when only the scenarios relating to an initial TAC of 800 t are considered (**Figure 106**). The catch rates respond in a very similar way to the spawning biomass, even though they, in fact, relate to the exploitable biomass. The catches, however, are more complex in their response (**Figure 107**).

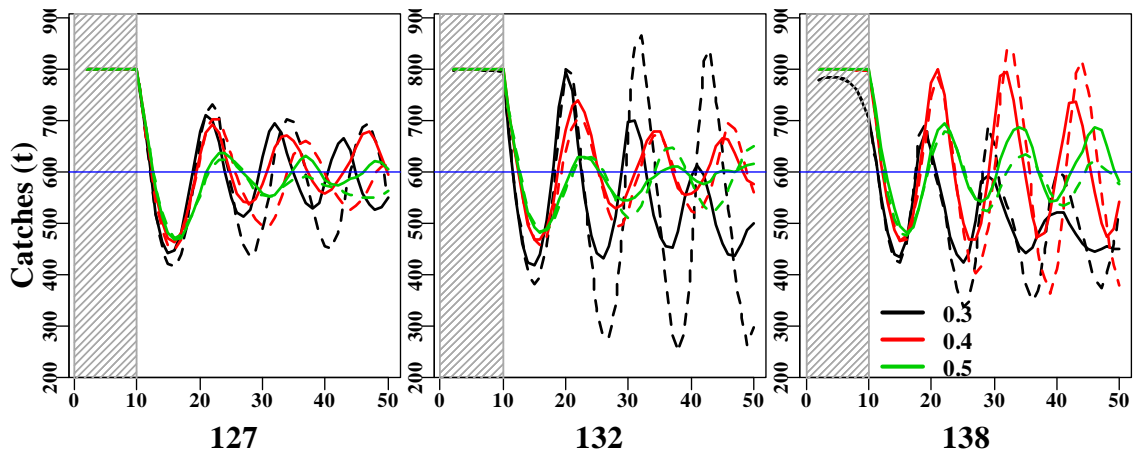


Figure 107. A direct comparison of the effects on the catches, which generally reflect the TACs, of different initial depletion levels (see legend) when the three different LML have an initial TAC of 800 t. The fine blue line is at 600 t to aid comparisons.

At an LML of 127mm the three initial depletion levels all tend to a final catch that is oscillating around about 600 t. At 132mm and 138mm LML, depletion levels of 50% and 40% appear to be stabilizing around 600 t, but the initial depletion of 30% leads to gradually declining catches with the oscillations being greater at the 132mm LML. With a TAC limit in the 132mm LML the catches oscillate much more widely than in the other scenarios, suggesting that the HCR is failing to respond quickly enough to changes.

Some the rates of change in TAC from year to year remain relatively high. While the option of a TAC limit was considered the notion of a limit on the proportional change on the TAC change was not explored. This might also prove of use in the context of multi-criterion decision analysis management framework and should be considered for future work. The effect of the period over which the catch rates are averaged does have an influence on the outcome but mainly on the variability of the fishery rather than the final depletion level or CPUE. Total catches can be slightly higher with CPUE period of 5 years because the HCR begins operation a few years earlier than the alternatives of 8 and 10 years (**Figure 108**).

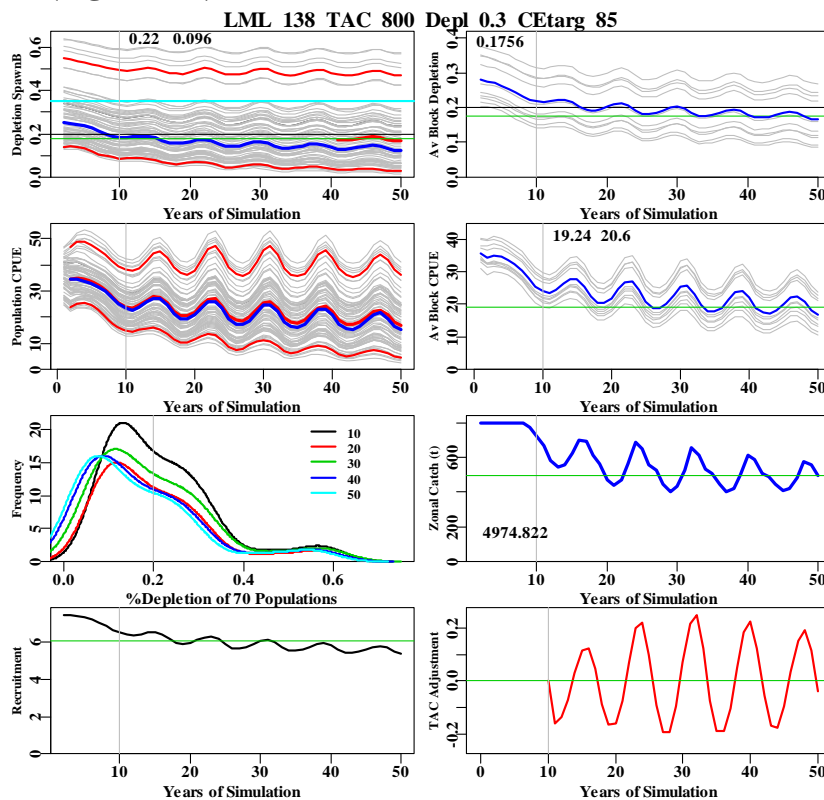


Figure 108. A single zone simulation with an initial depletion level of 0.3, an initial TAC of 800 t, and a LML of 138 mm. The HCR is the CE gradient but with a 5 year period, which begins operation in year 11.

With a CPUE period of 8 years there is only time for 1.5 fewer oscillations than with a period of 5 years (**Figure 109**). Despite the change in average dynamics the impact on the final depletion level is minor as it is on the final CPUE. Finally, with a CPUE period of 10 years, this delays the start of the introduction of the HCR by a further two years so it only starts in year 16. In all cases of 5, 8, or 10 years the final depletion is very similar as is the final CPUE. Variation in catches happens most slowly with the 10 year period but in all cases the range of changes in catches is very similar.

The effect of slow depletion over a large number of years is easily apparent in the pseudo-deterministic simulations. The oscillations in CPUE occur even though the spawning biomass does not change greatly, although the CPUE is at a very low level, so the absolute variation in CPUE is only of the order of 10 kg/hr.

The performance of this HCR does not recommend itself for use in management as it fails to encourage a recovery from depletion and maintains the status quo even when the stock could easily be fished more intensively.

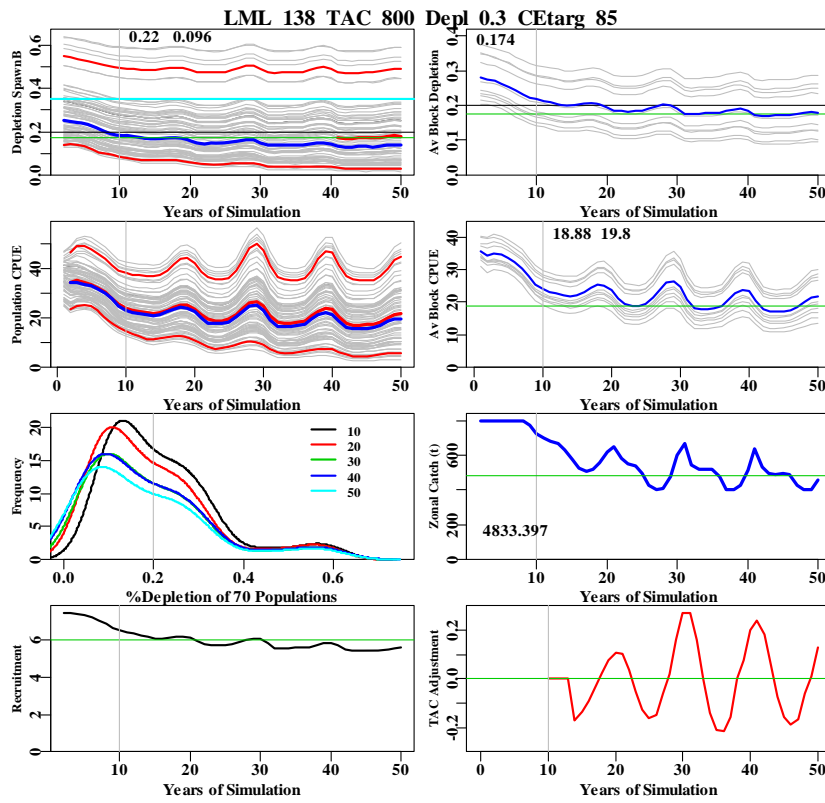


Figure 109. A single zone simulation with an initial depletion level of 0.3, an initial TAC of 800 t, and a LML of 138 mm. The HCR is the CE gradient but with an 8 year period, which begins operation in year 14.

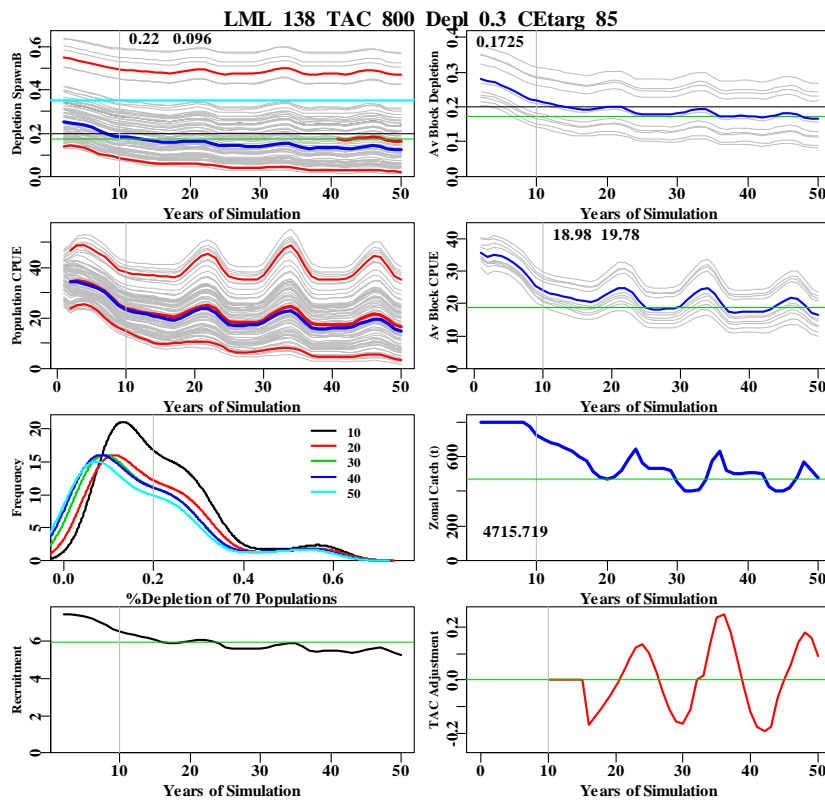


Figure 110. A single zone simulation with an initial depletion level of 0.3, an initial TAC of 800 t, and a LML of 138 mm. The HCR is the CE gradient but with a 10 year period, which begins operation in year 16.

21.4.1.1 Mean Shell Length

There is a possibility that extra information, in the form of catch length frequency data may have value in avoiding the seemingly inevitable decline brought on by the time lags built into the abalone stock dynamics and the CPUE gradient HCR. However, within any scenario there appears to be little contrast in the observed length frequencies of the catch with it varying by only about 4 mm, which would be very hard to detect given natural variation and limited sample sizes (**Figure 111**). Despite this apparent lack of information, given the differences apparent between different levels of depletion this performance measure would be a candidate for further investigation that included multiple performance measures in one management framework.

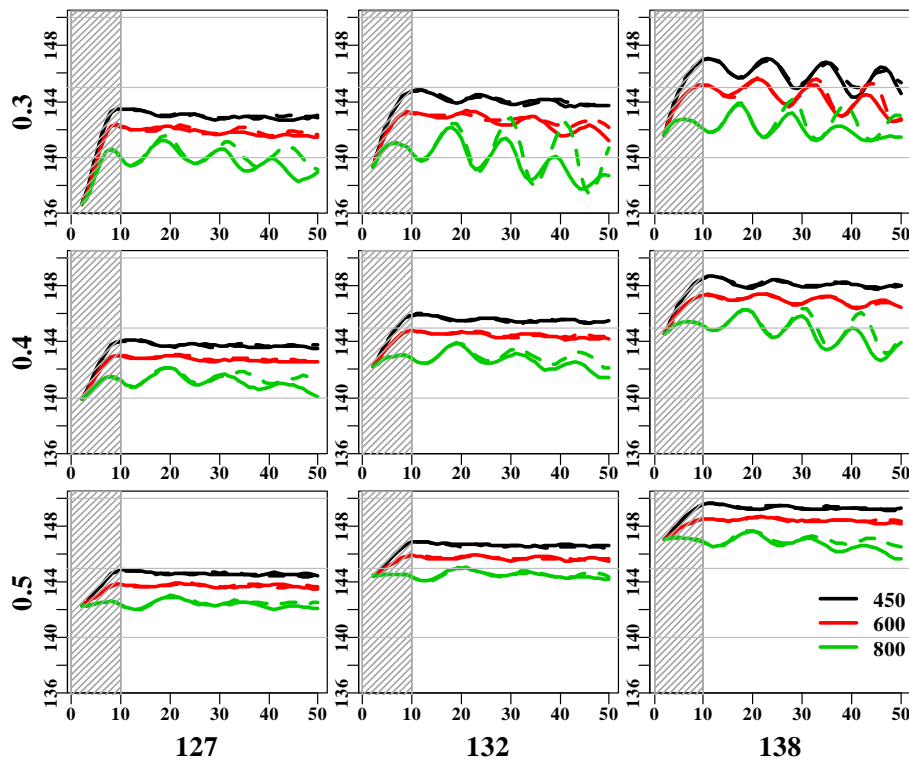


Figure 111. Mean Shell Length (mm) under the dynamics of the 45 different scenarios of initial depletion (y-axis), LML (x-axis) and initial TAC considered with the Gradient CPUE HCR. The dashed lines relate to where no lower limit is imposed on the TAC and the grey box is the period of initiation.

21.4.2 Gradient CPUE with Initial TAC Perturbation

With experience from age structured models a variant to the CPUE gradient HCR was considered where the TAC was perturbed at the initiation of the HCR. In this case the TAC was reduced by 25% in the first year.

The implementation of a perturbation improved the final depletion levels, the total catch and the average catch rates but only when the stock wasn't depleted beyond 40% initially (**Figure 112** and **Figure 113**). If initially depleted to 30%, then the perturbation had very little effect, primarily because the HCR immediately led to an equivalent reduction in TAC as the perturbation. The same conclusion arises that the HCR failed to achieve useful management of the abalone fishery (**Figure 114** and **Figure 115**).

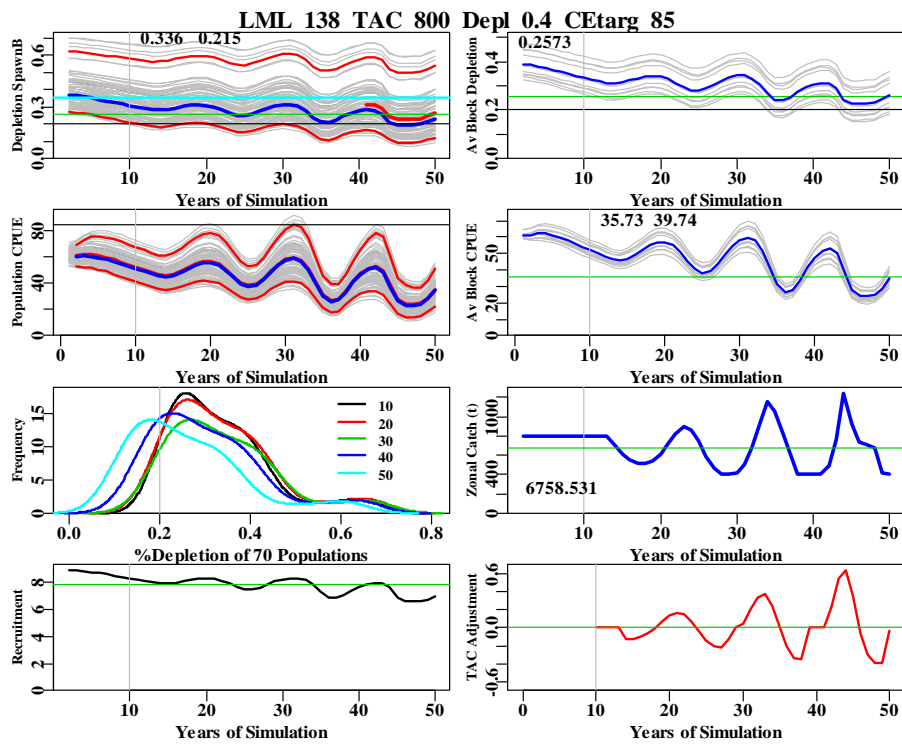


Figure 112. A single zone simulation with an initial depletion of 40%, an initial TAC of 800 t, and a LML of 138 mm. The HCR is the CE gradient but with an 8 year period and no initial perturbation to the TAC.

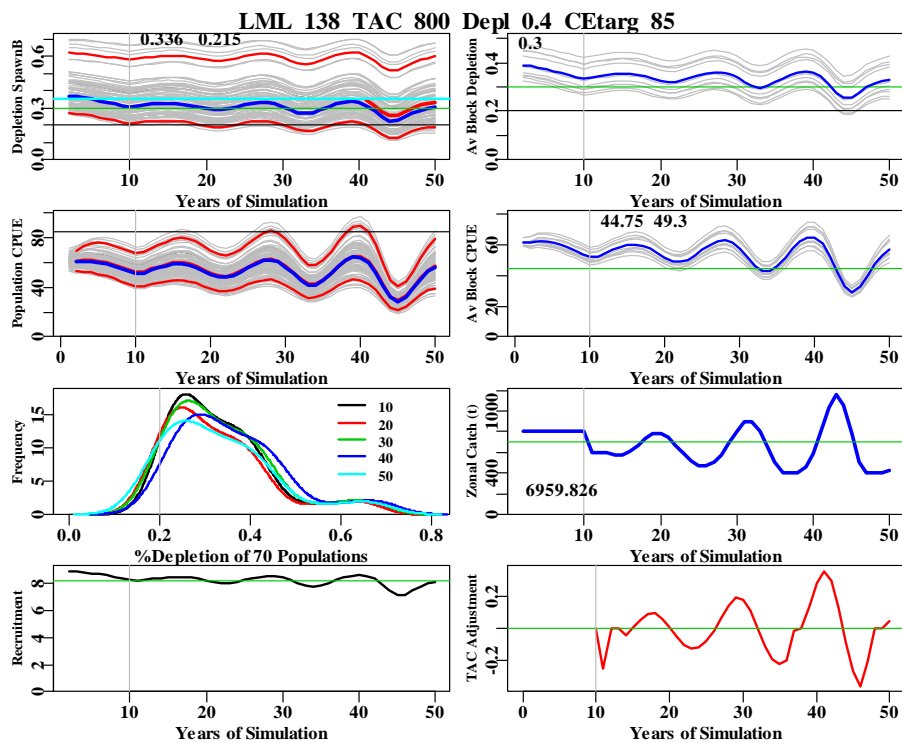


Figure 113. A single zone simulation with an initial depletion of 40%, an initial TAC of 800 t, and a LML of 138 mm. The HCR is the CE gradient but with an 8 year period and an initial perturbation to the TAC of a 25% reduction in year 11.

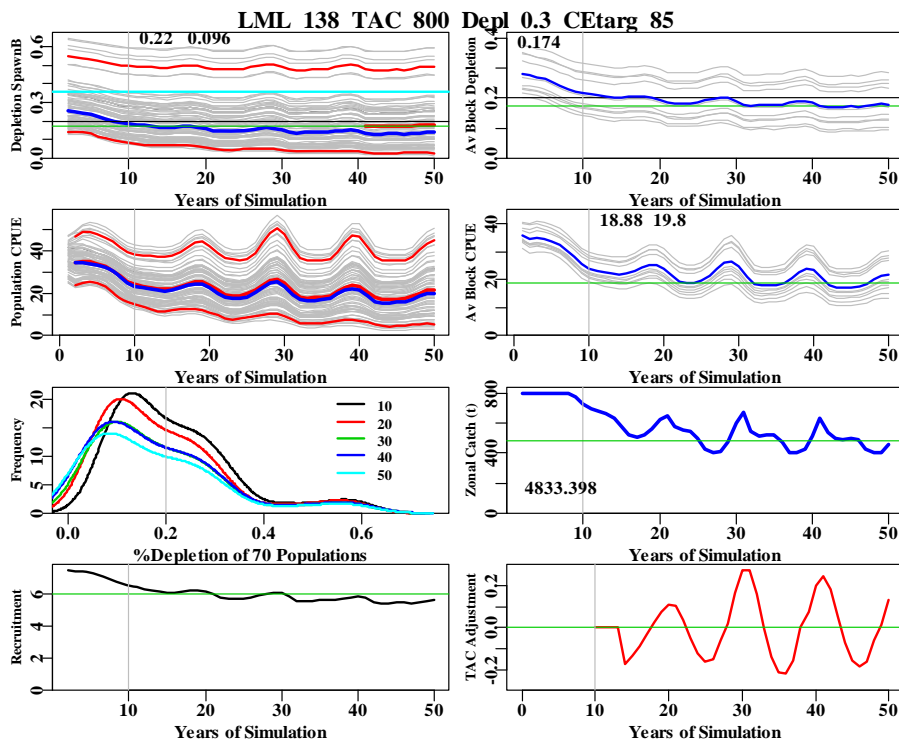


Figure 114. A single zone simulation with an initial depletion of 40%, an initial TAC of 800 t, and a LML of 138 mm. The HCR is the CE gradient but with an 8 year period and without an initial perturbation to the TAC.

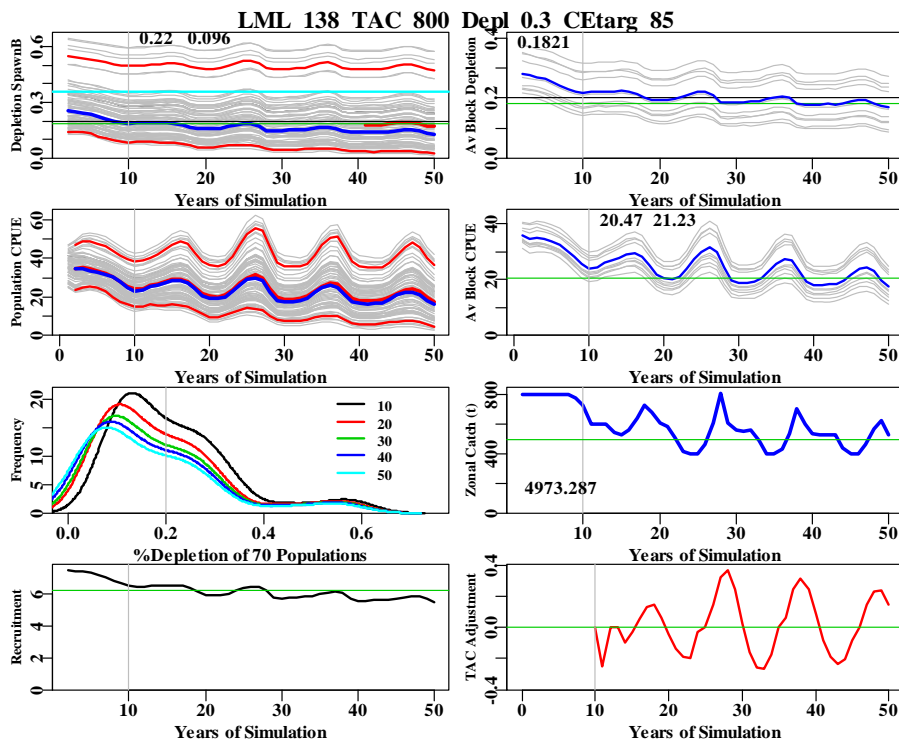


Figure 115. A single zone simulation with an initial depletion of 30%, an initial TAC of 800 t, and a LML of 138 mm. The HCR is the CE gradient but with an 8 year period and an initial perturbation to the TAC of a 25% reduction in year 11.

21.4.3 Asymmetric Gradient CPUE HCR

In the CPUE gradient HCR examined here the response of the TAC to changes in the gradient of CPUE has been symmetric, meaning that the TAC is increased and decreased by the same degree for both positive and negative gradients; this reflects current practice with CPUE (not CPUE gradients) in South Australia (see Chick and Mayfield, 2012, p 14). However, if asymmetry is introduced into the response such that positive gains in TAC in response to positive gradients are less than negative losses to the TAC in response to negative gradients, then this modified HCR appears capable of recovering an initially depressed stock (**Figure 116**).

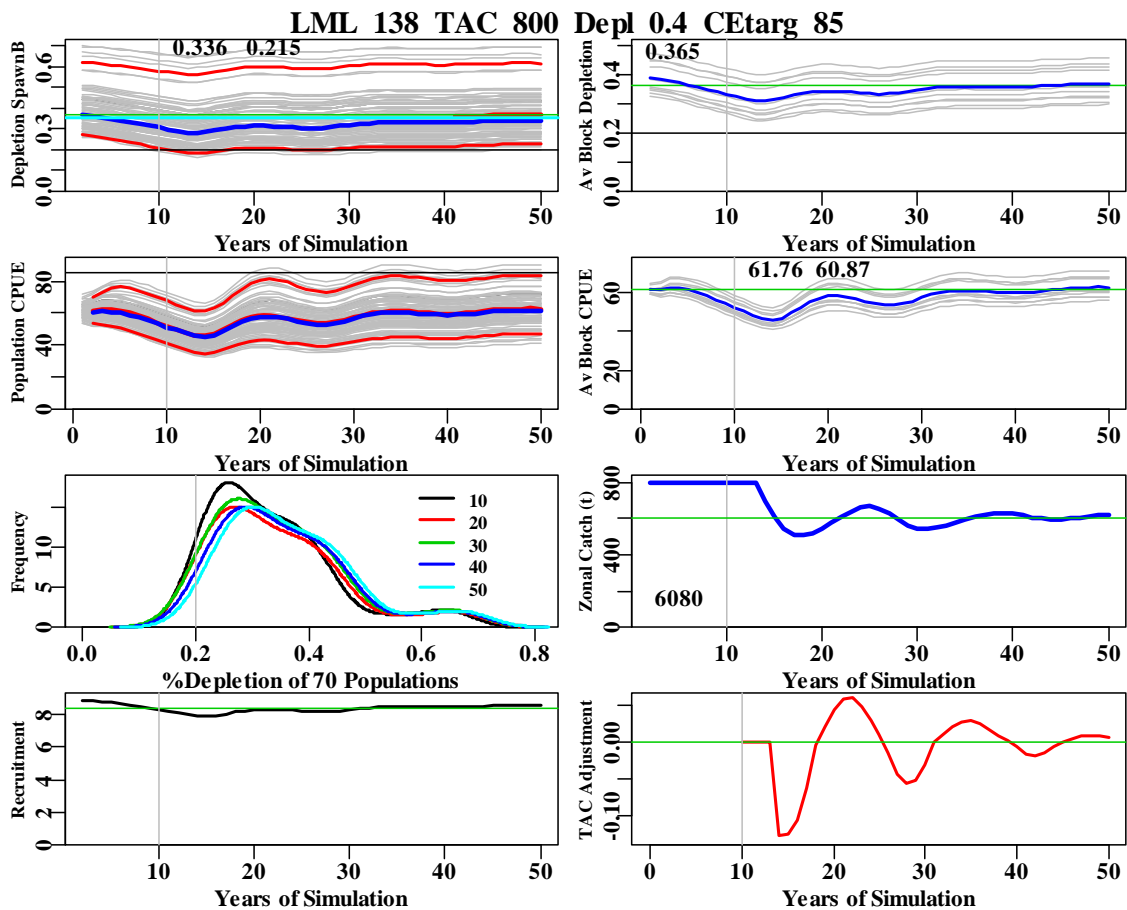


Figure 116. The application of a modified CPUE gradient HCR where the TAC response is asymmetric such that increases are smaller than decreases. In this case, the modification was to divide positive gradients by 3.0 but to leave negative gradients unchanged. Once included in equ (86) this had the effect of introducing the required asymmetry of response.

The rate of recovery is greatly influenced by the degree of asymmetry. It requires increased asymmetry to greater the level of depletion. Otherwise the modification is ineffective and no recovery occurs. Nevertheless, this appears to be a positive way forward that might still provide value from estimating the gradient of recent CPUE.

21.4.4 Target CPUE HCR

The target HCR examined used a target CPUE of 85 kg.hr, which was about 5 kg/hr above the catch rates expected at the potential maximum yield (**Table 41**). Scenario combinations included: four initial TACs (450, 600, 750, and 800 t; although in most results only 450, 600, and 800 are considered for contrast), three initial depletion levels (0.3, 0.4 and 0.5 B_0), three LML (127, 132, and 138 mm), and there was either a TAC limit of 0.5 the initial TAC or there wasn't. This produced 72 or 54 combinations depending on whether the 750 t TAC was considered.

Table 41. Empirical estimates of maximum sustainable yield from a simulated zone with a BML of 138 mm. iTAC is the initial TAC, iDepletion is the initial spawning biomass depletion, SumC is the sum of catches from years 16 – 50 (35 years), while the final depletion and CPUE are the averages over years 16 – 50. The eMSY is the empirical MSY which is SumC divided by 35 (e.g. **Figure 117**). All analyses were conducted on a simulated zone with a BML of 138mm so care is required with interpretation.

LML	iTAC	iDepletion	SumC	eMSY	Final Deletion	Final CPUE
127	640	0.30	22400	640	32.21	76.04
132	645	0.38	22575	645	37.53	80.49
138	630	0.44	22050	630	44.51	80.45

21.4.4.1 Empirical MSY Predictions (eMSY)

By setting the stochasticity in the fishery dynamics (diver estimates of exploitable biomass and observed CPUE) along with recruitment variability to extremely low values ($1e^{-6}$) the simulated dynamics becomes effectively deterministic. In that way the initial conditions that lead to stable and maximal production under the different scenarios can be determined (**Table 41**, **Figure 117**).

The outcomes from the diagnostic statistics in the following analyses can be compared to these ideal outcomes (**Table 41**, **Figure 117**). The unfished spawning biomass is the same in all cases. A major difference is that the 138mm LML maximizes production at 44.5% of spawning biomass at eMSY, whereas the 127mm LML needs to deplete the resource to only 32.2%, the catch rates remain very similar at the maximum productivity, ranging from 76 – 80.5 kg/hr, and the eMSY only varies from 630 – 645 t. The potential benefits of selecting a LML of 138mm if the BML is 138 mm relate primarily to the stock having greater resilience to unexpected perturbations through having a higher standing biomass rather than increased yields or higher catch rates.

In each case with no stochasticity the zone-wide catches and TACs were all unchanged through the simulation but the depletion and catch rates were very slowly declining. The slow declines in depletion and CPUE were more pronounced in the 127 and 132mm LML cases, but even their reductions were relatively small (**Figure 117**). The outcomes were not flat because the resolution of the Target CPUE HCR would not allow finer adjustments to the initial values given only minor changes in catch rates through time.

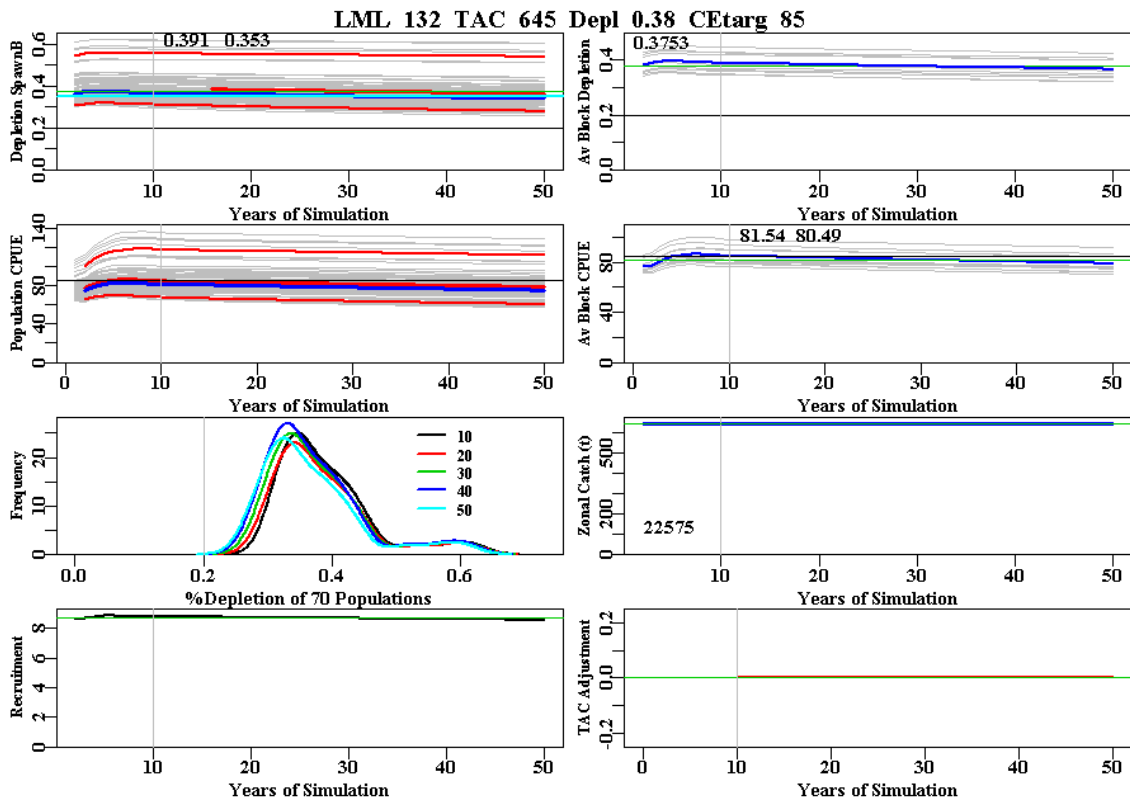


Figure 117. A single zone simulation run with stochasticity set to very low values to illustrate almost deterministic behaviour. There is a close match between the initial TAC and the initial depletion level. The spawning biomass and CPUE are not parallel with the x-axis but within the precision of operation of the HCR this is stable.

21.4.5 Trajectory Plots for Alternative LML

The performance of the target CPUE HCR can be illustrated relative to the target chosen. The particular trajectory followed on average, across the 100 replicate zone simulations, varied depending on the starting conditions. The different LML in various combinations of initial conditions led to trends of change in the trajectories exhibited by the catches, CPUE, and spawning biomass; these trends were also modified depending on whether or not each scenario included a limit on the lowest TAC. By considering the effect upon the spawning biomass, the catch rates, and the catch (TAC) at the same time, across the range of initial TAC and initial depletion levels, the interactions between these two inputs and how they interact with the limit on the lowest TAC permitted can be elucidated (**Figure 118**, **Figure 119**, and **Figure 120**).

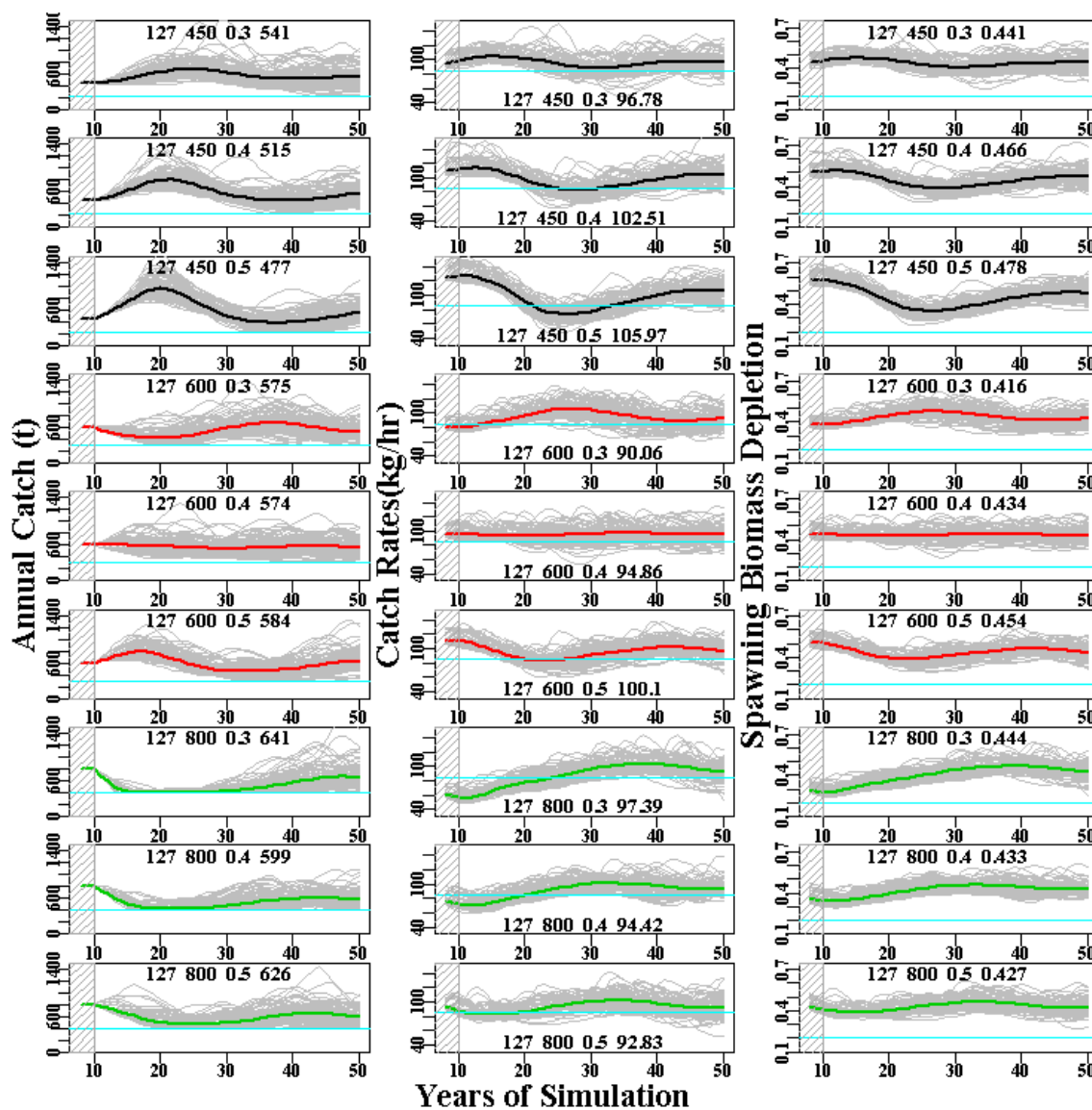


Figure 118. The annual catch (TAC), the catch rates, and the spawning biomass depletion level through the years of the simulation for all 9 combinations of the three initial TAC and three initial depletion levels for an LML of 127 mm with a TAC limit turned on. The annotation on each graph is in each case the LML, the initial TAC (iTAC), and the initial depletion level. For each column the last number in each case is the average over the final ten years, for catches, catch rates, and spawning biomass depletion levels respectively. In each case, for reference, the pale blue line is, by column, the TAC limit ($= 0.5 \times \text{iTAC}$), the target catch rate, and $20\%B_0$. Note the y-axes for catch rates and spawning biomass depletion do not start at zero.

Within each scenario the pattern of response of the catch rates and the spawning biomass were very similar, which merely illustrates that the exploitable biomass tends to follow the same trajectory as the spawning biomass (although the ratio of exploitable to spawning biomass varies with LML). The pattern of response of catches to the initial TAC and initial depletion level was similar under each of the three LML although the depth of any declines and the heights of any rises increased with LML. As expected, given the empirical MSY values the initial TAC of 450 t values tend to lead to relatively high spawning biomass levels and related catch rates at the end of the initiation period, which in turn lead to increases in catches; although exceptions occurred when an iTAC of 450 t was combined with the two larger LML and lowest initial depletion levels. The increased catches are initially associated with increases in catch rates and spawning biomass which then fall after a time lag by which time catches have once more increased beyond surplus production. Surprisingly, with the initial TAC of 450 t, the 50% initial

depletion level led to the most variable response in biomass, catch rates and catches. This again reflected time lags in the response of catch rates to changes in the catches, which derive from the time it takes for new recruits to grow from the spawning biomass the availability of which depends eventually upon the catch levels.

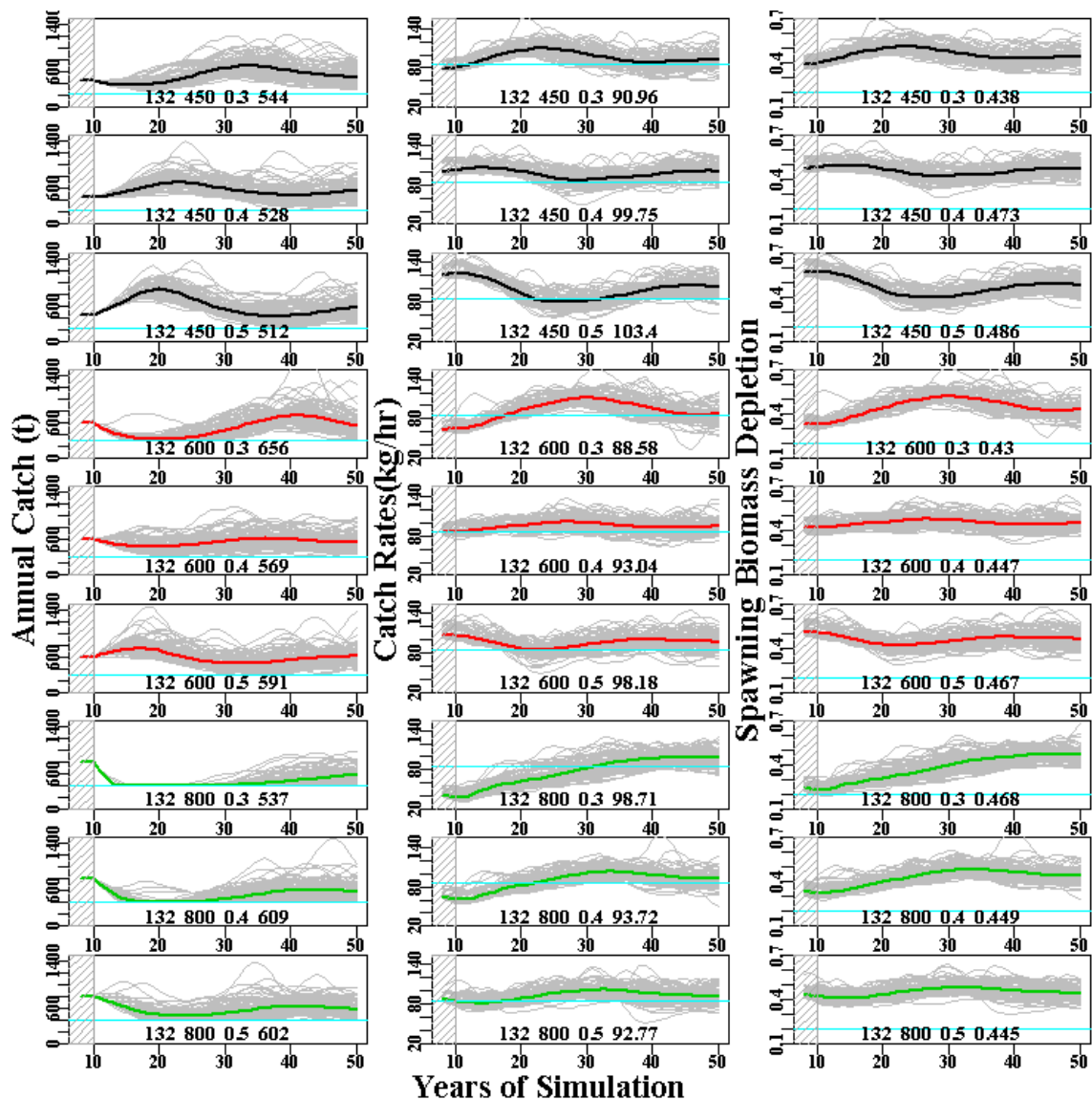


Figure 119. The annual catch (TAC), the catch rates, and the spawning biomass depletion level through the years of the simulation for all 9 combinations of the three initial TAC and three initial depletion levels for an LML of 132 mm with a TAC limit turned on. The annotation on each graph is in each case the LML, the initial TAC, and the initial depletion level. For each column the last number in each case is the average over the final ten years, for catches, catch rates, and spawning biomass depletion levels respectively. In each case, for reference, the pale blue line is, by column, the TAC limit ($= 0.5 \times \text{initial TAC}$), the target catch rate, and $20\%B_0$. Note the y-axes for catch rates and spawning biomass depletion do not start at zero.

The most stable trajectories are exhibited with the initial TAC of 600 t. With the LML of 127mm the 40% initial depletion level remains generally flat, with a final ten year mean of 43.4% at an average catch of 574 t. Events were slightly more variable with the LML of 132mm but that still exhibited a final 10-year mean of 44.7% depletion and average catch of 569 t. With the 138mm LML a similarly stable trajectory was only produced by an initial depletion of 50% which ended with a final ten year mean of 48.8% and an average catch of 566 t.

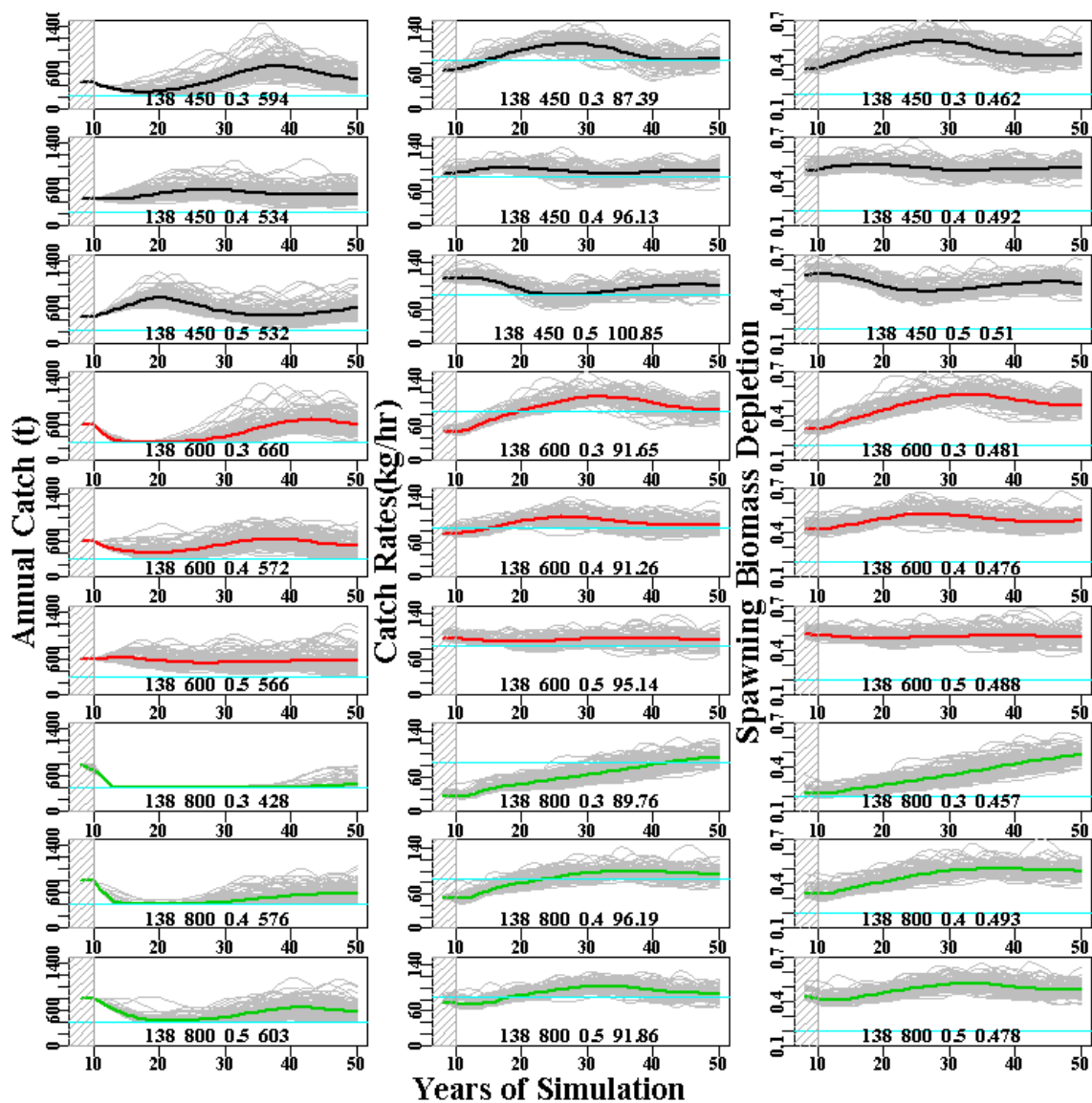


Figure 120. The annual catch (TAC), the catch rates, and the spawning biomass depletion level through the years of the simulation for all 9 combinations of the three initial TAC and three initial depletion levels for an LML of 138 mm with a TAC limit turned on. The annotation on each graph is in each case the LML, the initial TAC, and the initial depletion level. For each column the last number in each case is the average over the final ten years, for catches, catch rates, and spawning biomass depletion levels respectively. In each case, for reference, the pale blue line is, by column, the TAC limit ($= 0.5 \times \text{initial TAC}$), the target catch rate, and $20\%B_0$. Note the y-axes for catch rates and spawning biomass depletion do not start at zero.

21.4.6 The TAC Limit Option

The effect of the lower limit on the TAC is primarily exhibited by the 800 t initial TAC scenarios and any effects appear to be exacerbated by increases in the LML. Thus, with the initial TAC of 600 t, there is a slight influence at an initial depletion of 30% and LML of 127mm, and the scale of the impact increases as the LML increases to 132mm and 138mm (**Table 42**). At the 800 t iTAC an influence on catches is observed at all initial depletion levels but the intensity and duration of that impact increases as the LML increases (**Table 42**). This has consequent effects on the spawning biomass and on catch rates. While the impacts of the lower limit on TAC are illustrated in (**Figure 118**, **Figure 119**, and **Figure 120**) they are more clearly illustrated using histograms

that identify the first year of hitting the limit TAC (**Figure 121, Table 42**) and another illustrating the duration of being at the limit (**Figure 122**).

Table 42. Counting the number of years from 11 – 50, across the 100 replicates, where the TAC hits the TAC lower limit of 0.5 x TAC. The rows are the combinations of TAC and Depletion and the columns are the different LML with columns for the actual counts (out of 4000) of the total number of years at the limit and the equivalent percentages. This only relates to those scenarios where a TAC limit was in place.

Scenario		Counts			Percentage		
		127	132	138	127	132	138
450	0.3	5	0	14	0.125	0	0.350
450	0.4	0	0	0	0	0	0
450	0.5	20	0	0	0.500	0	0
600	0.3	16	481	1135	0.400	12.025	28.375
600	0.4	5	2	4	0.125	0.050	0.100
600	0.5	51	17	7	1.275	0.425	0.175
800	0.3	1599	2292	3108	39.975	57.300	77.700
800	0.4	883	1208	1632	22.075	30.200	40.800
800	0.5	425	410	772	10.625	10.250	19.300

The patterns visible in **Figure 118, Figure 119**, and **Figure 120** with respect to the rapidity in which the TAC limit was reached in each case become more apparent when counts are made of the actual years in which the limit is reached (**Figure 121**). The difference between the 600 t and 800 t initial TACs become very clear, it also becomes clear that the higher the LML for both TACs the earlier and more concentrated is the year in which the limit is reached.

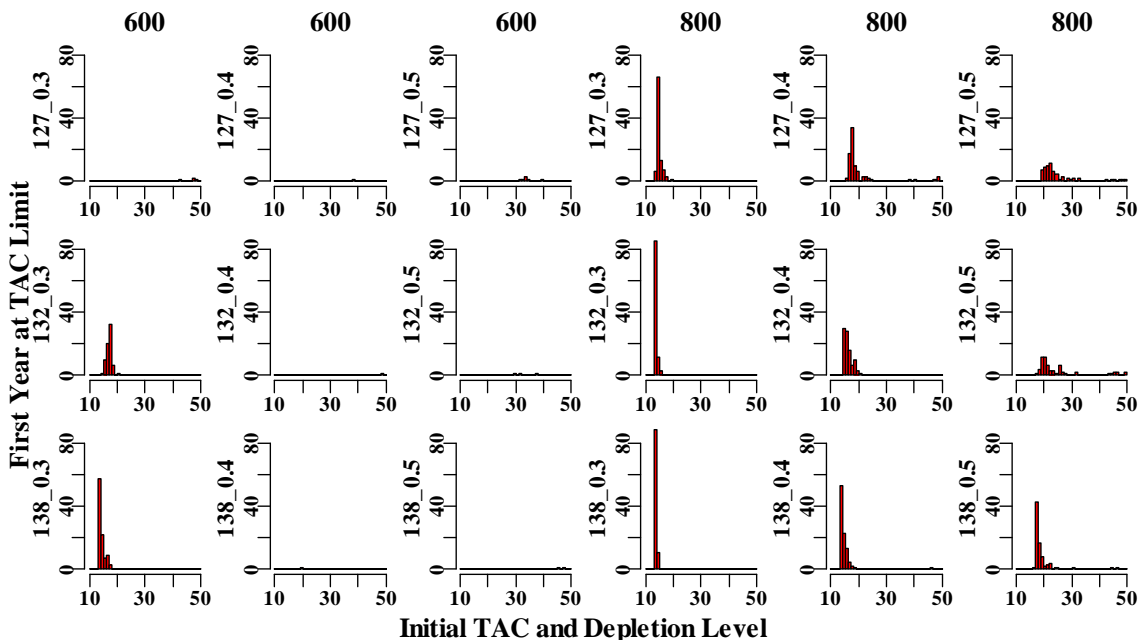


Figure 121. Frequency counts of the different years in which the TAC reaches the TAC limit under different scenarios in which the limit is implemented.

The number of years for which the TAC is held at the limit also becomes clearer. For the iTAC of 600 and depletion of 30%, and all the iTACs of 800 t, as the LML increases the mode of the distribution of periods below the TAC limit moves across to a higher number of years.

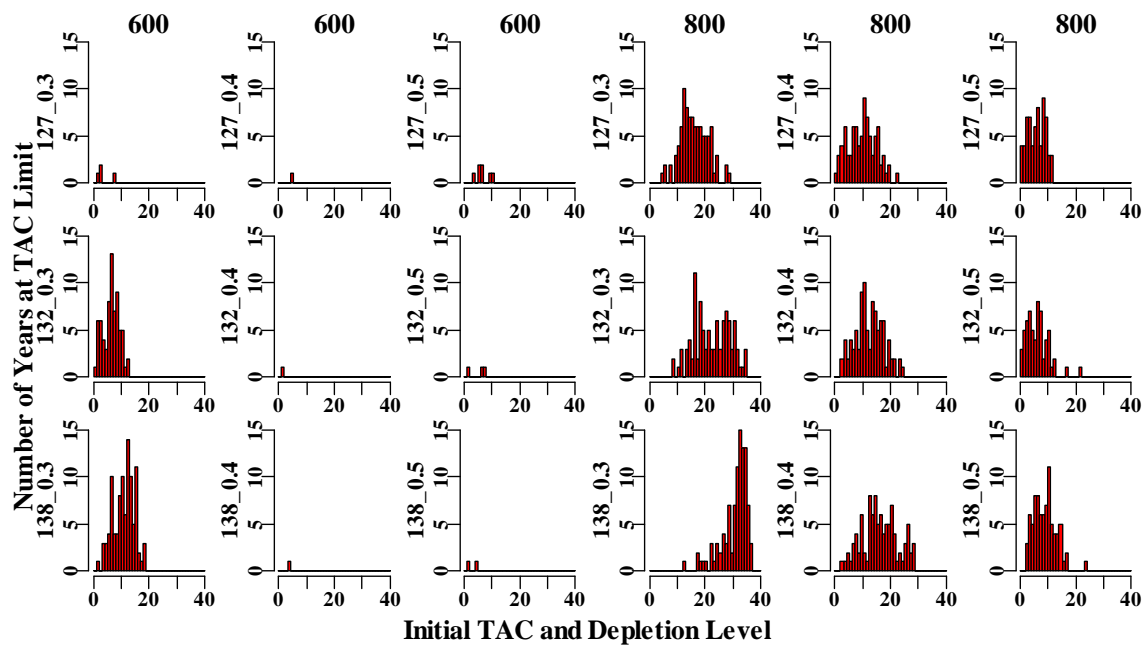


Figure 122. Frequency counts of the number of years in which the TAC is at the TAC limit under different scenarios in which the limit is implemented.

In all scenarios with a 138mm LML and an iTAC of 800 t the introduction of a TAC limit reduces the overall variability in the response the HCR exhibited in all scenarios considered (**Figure 123**). Including a TAC limit slows the recovery by increasing the number of years for the CPUE to attain the target for the first time, and it also reduces the average catch in the final 10 years although it increases the total catch across the final 35 years. It does this through the higher catches brought about by the lower limit being placed in the TAC leading to a slower recovery of the spawning biomass and related exploitable biomass. In the extreme case of 800 t and 30% initial depletion, with a TAC limit it takes almost 2.5 times as long (32 years) for the average CPUE across replicates to achieve the target than when no limit is in place. In addition, in at least one replicate the recommended TAC gets as low as 56 t (from a high of 800 t). There is clearly a trade-off between average catch and years to recover, however, related to that is the difference that can be expected in the catch rates. The smaller catches when there is no TAC limit will also lead to higher catch rates under that version of the HCR (**Table 43**).

Table 43. Years to reach the CPUE target, and Catch variability in the 138mm LML and 800 t iTAC scenarios. Years is the number of years it takes the average CPUE across each set of 100 replicates to meet the CPUE target of 85kg/hr. AvC is the average catch over the final 10 years, SumC is the total catch, and MinC and MaxC are the range of catches across the final 35 years.

Limit	iDepletion	Years	AvC	SumC	MinC	MaxC
T	0.3	32	428	14299	400	779
F	0.3	13	698	11274	56	1386
T	0.4	13	576	16976	400	1067
F	0.4	9	696	16582	171	1309
T	0.5	9	603	18359	400	1147
F	0.5	9	625	18332	269	1256

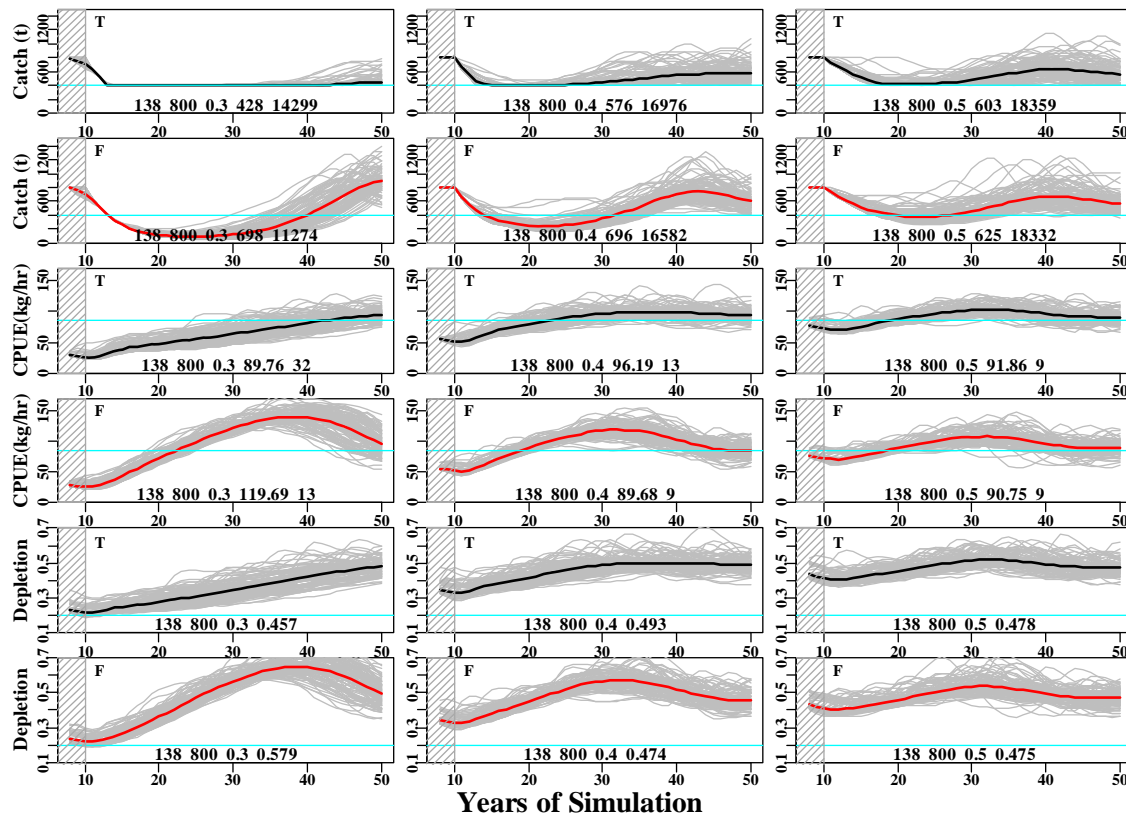


Figure 123. A comparison of the effect of the TAC limit (black lines and ‘T’ superscript) with the same scenarios run without a TAC limit (red lines and ‘F’ superscript). In all scenarios only an initial TAC of 800 t and an LML of 138mm was considered. The last two numbers in the catch graphs are the average catch over the last 10 years and the total catch across the last 35 years. In the CPUE graphs they are the average CPUE over the last 10 years and the number of years from year 10 for the target CPUE to be reached.

The potential trade-off between total catch and CPUE within a set of years is a potentially important question but, unfortunately this only tends to become an issue when a stock is in a depleted state and the fishery is reduced. Keeping catches elevated is a riskier option (TAC Committee, 2012).

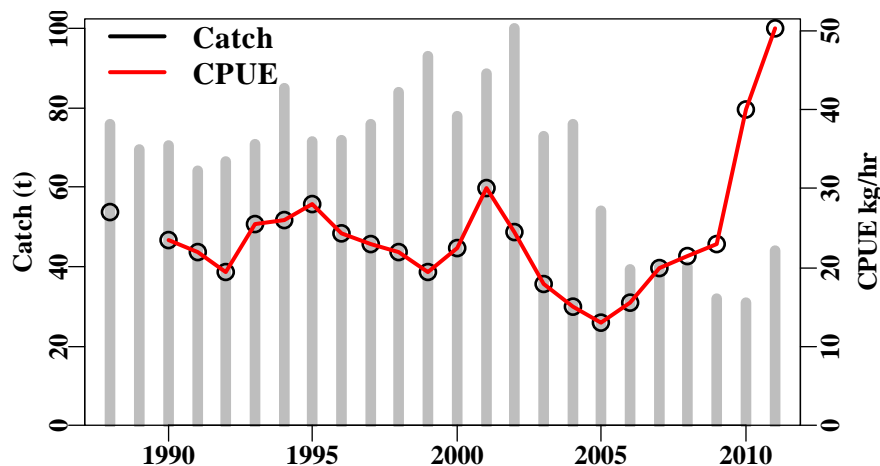


Figure 124. Approximate catches (bars) and CPUE (red line) from 1988 – 2011 in Region 6 (the southernmost area) in the NSW abalone fishery. The CPUE clearly increased dramatically following large TAC reductions (data approximated from a graph in TAC Committee, 2012).

If the limit TAC is set too high it could prevent a stock recovering. For example, by setting up a stochasticity free single zone simulation with an iTAC of 800 t, an iDepl of 30% and a LML of

138mm, and with a limit TAC of $0.675 \times \text{iTAC}$ (rather than $0.5 \times \text{iTAC}$ as used in the simulations above), this was sufficient to generate an essentially stable outcome with ongoing depressed catch rates (**Figure 125**). This state of low level stable catch rates is reminiscent of the situation that was being expressed in NSW, Australia until TACs were cut to relatively low levels following which there are signs of recovery and CPUE levels unseen for decades (TAC Committee, 2012).

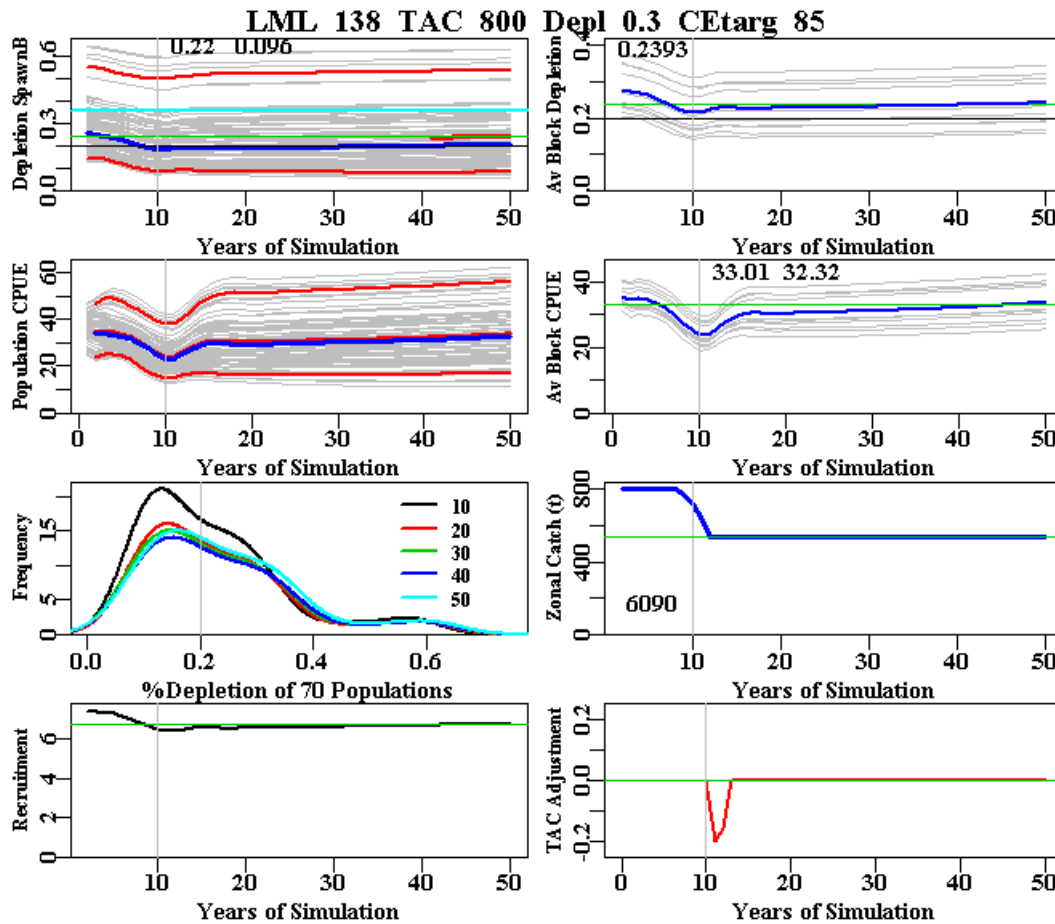


Figure 125. A single zone simulation with stochasticity set to $1e^{-6}$ with an iTAC of 800 t, an iDepl of 30% and a LML of 138mm, with a limit TAC of $0.675 \times \text{iTAC}$.

21.4.7 Years to Achieve the CPUE Target

In the previous section the years to attain the CPUE target were estimated using the average catch rate across all replicates within a scenario. A better estimate can be obtained by sorting through each replicate for each scenario and recording the first year the CPUE target is attained. From this the 100 estimates can be summarized into tables (**Table 44**) and figures (**Figure 126**) accordingly. In this way the time to reach the target can be identified.

All instances where the mean year to target was 1 with a standard deviation around that of 0 represent scenarios where the target was met before the end of the 10 year initiation period. Mostly this occurred where the iTAC was 450t combined with a iDepl of 40 or 50% all scenarios achieved the target before the end of the initiation period of 10 years. This also occurred with an iTAC of 600t combined with an iDepl of 50. However, there were a few other instances where achieving the target before year 10 out of the 50 occurred (**Table 44**).

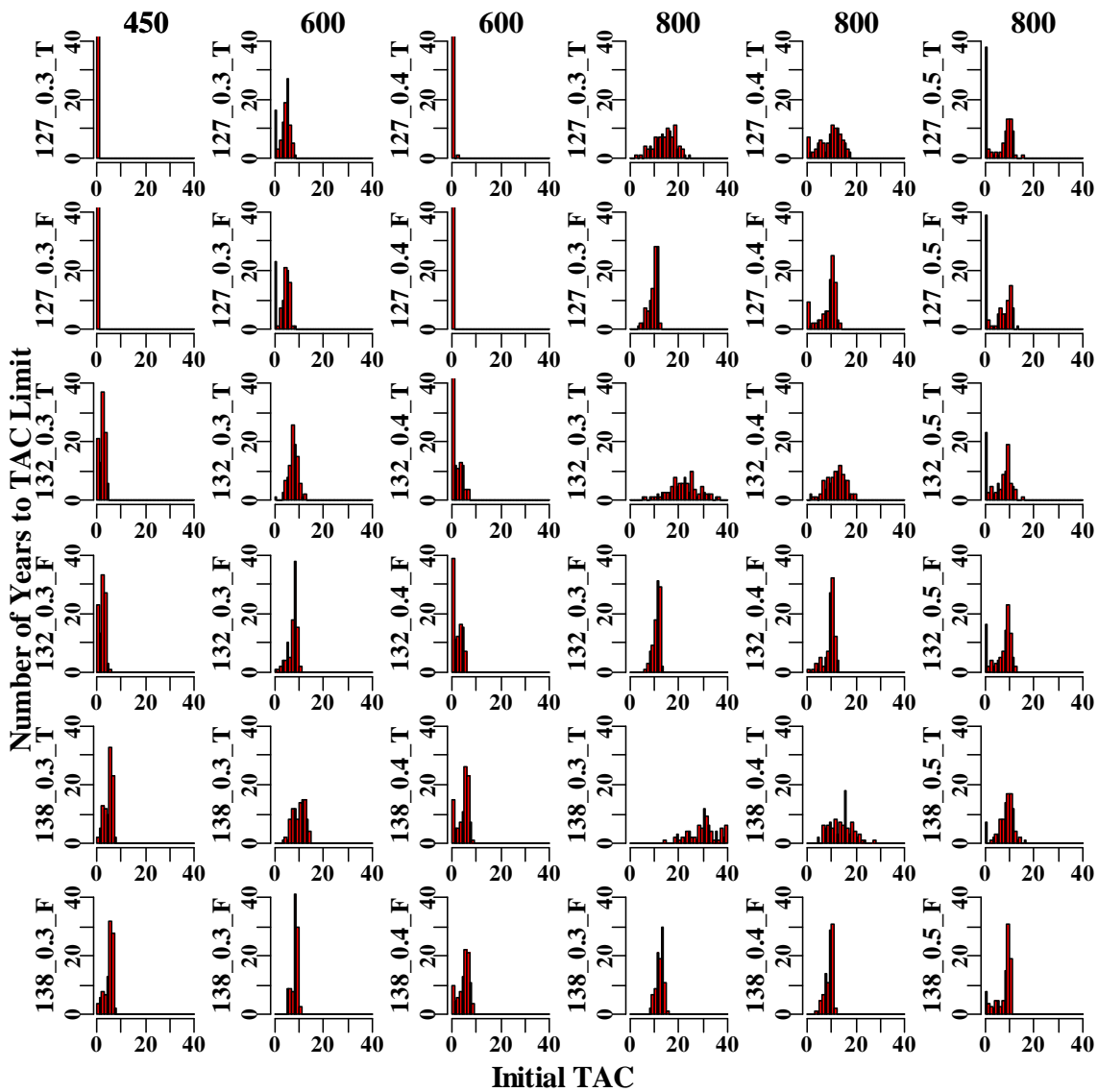


Figure 126. The number of years (from year 11 out to 50) for each simulated zone to reach the target catch rate of 85 kg/hr. Each graphs y-axis label relates the LML, the initial depletion and whether there was a TAC limit or not. The main column headings relate to the initial TAC for each scenario. The iTAC 450 with iDepletion of 40 and 50% and the iTac of 600t and iDepletion of 50% were omitted as all reaching the target before the end of the ten year initiation period. Where there is only a single column extending past 40 implying all records achieved the CPUE target before year 10.

Within any particular iTAC and iDepl level, as the LML increased the number of years it takes to reach the target increases. On top of this, if there is a TAC limit this also increases the number of years to reach the target (**Table 44, Figure 126**).

Only in the case of the iTAC of 800t, an iDepl of 30%, and a LML of 138mm were there instances where not all replicate mean CPUE values reached the CPUE target in the 40 years following the introduction of the Target CPUE HCR (**Table 44**).

Table 44. The mean number of years (from year 11 out to 50) for each simulated zone to reach the target catch rate of 85 kg/hr. The Limit TAC reflects whether there was a simulation recommended a TAC below which catches were not allowed to decline. The 127, 132, and 138 are the LML for each simulation. All statistics relate to 100 replicate simulation runs. The highlighted cell only attained the target in 83 out of 100 replicates.

Initial Depletion	Limit TAC	Initial TAC	127		132		138	
			Mean	StDev	Mean	StDev	Mean	StDev
0.3	TRUE	450	1	0	2.8	1.1892	5.24	1.6765
0.3	TRUE	600	4.71	2.1144	8.13	2.0134	10.61	2.5302
0.3	TRUE	800	14.89	4.3598	22.37	6.3955	30.72	5.7410
0.3	FALSE	450	1	0	2.77	1.2215	5.37	1.7561
0.3	FALSE	600	4.4	2.2202	7.98	1.9999	8.83	1.2875
0.3	FALSE	800	10.23	1.9272	11.59	1.4571	12.87	1.5351
0.4	TRUE	450	1	0	1	0	1	0
0.4	TRUE	600	1.02	0.2	2.65	1.8443	5.05	2.2669
0.4	TRUE	800	10.06	4.5545	12.22	3.8654	14.17	4.6407
0.4	FALSE	450	1	0	1	0	1	0
0.4	FALSE	600	1	0	2.78	1.7441	5.38	2.2283
0.4	FALSE	800	8.99	3.4917	9.86	2.2519	9.49	1.6174
0.5	TRUE	450	1	0	1	0	1	0
0.5	TRUE	600	1	0	1	0	1	0
0.5	TRUE	800	6.15	4.6371	6.77	4.1191	9.11	3.2811
0.5	FALSE	450	1	0	1	0	1	0
0.5	FALSE	600	1	0	1	0	1	0
0.5	FALSE	800	5.84	4.4466	7.55	3.7183	8.03	3.2113

21.4.8 Maintaining the Target CPUE

By comparing the catch rates across all scenarios (**Figure 127**, **Figure 128**) it is possible to see the effects of both the increase in LML and the interaction with the TAC limit across the different iTAC and iDepl levels. As seen before the overall variability of catch rates was greater when there was no limit on TAC than when there was and this variability increased with LML. The effect on variability of different initial depletion levels interacted with the iTAC. Thus, both with and without a TAC limit, the most variable catch rate series for the iTAC of 450 t came at an iDepl level of 50% while that for the iTAC of 800 t was with the iDepl of 30%. That for the iTAC of 600 t was mixed with relative stability across the 40% iDepl level scenarios and similar variation between the 30 and 50% iDepl for the smallest LML shifting to greatest variation at the iDepl of 30% for the 138 mm LML.

An important feature of the catch rate trajectories through time is the occurrence of long term oscillations with rapid changes occurring across time frames of between 5 – 10 years (**Figure 127**). This is different from those seen on the east coast of Tasmania where whole cycles of up and back down again can occur over a 10 year period (Tarbath and Gardner, 2012). While this suggests the recruitment dynamics in the scenarios here are incorrect, the general trends and reasons behind them remain valid.

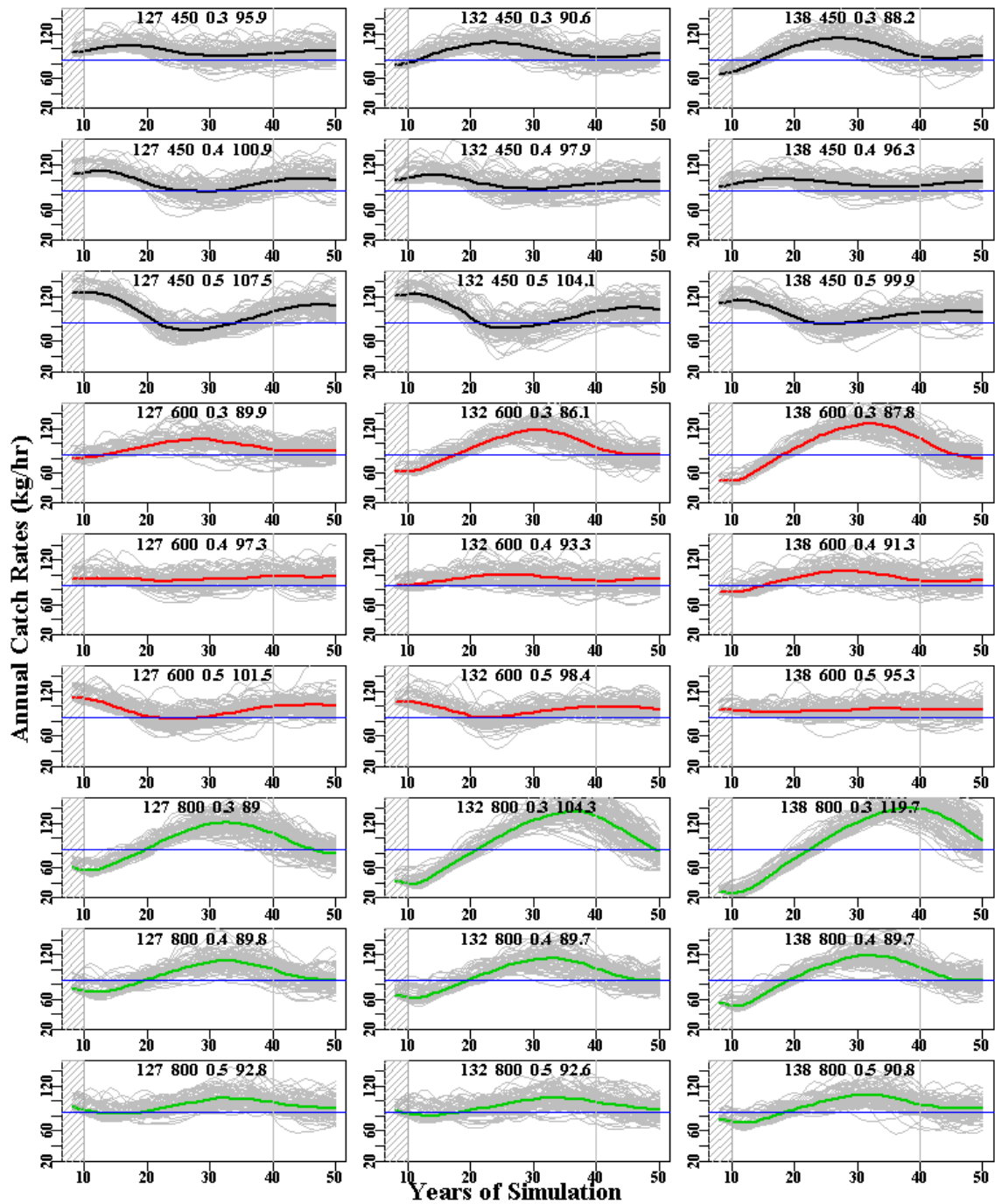


Figure 127. Envelope plot trajectories of the catch rates under each scenario where the TAC limit was not introduced. Each graphs header includes the scenario but also the average catch rate over the final 10 years. The blue line in each case is the target and the grey vertical line identifies the year 40 from where the final average CPUE is estimated.

While the CPUE target in the scenarios considered was set at 85 kg/hr, many of the outcomes fail to stabilize at that value. The relatively stable scenarios were those with an iTAC of 600 t and an iDepl of 40% (between 40 – 50% for the 132mm LML and 50% for the 138mm LML (**Figure 127**). None of these, with or without the TAC limit get within 6 kg/hr of the target and all appear to be biased high, with some being 10 kg/hr off (**Figure 127**, **Figure 128**). Given the relative stability of these scenarios this does not appear to be related to the time lags inherent in using CPUE to guide the HCR. Rather the problem appears to reflect a property of the algorithm inside the multi-criterion

decision analysis. By using the function ‘trunc’ in its definition, see equation (90), this invariably drops the score of each SAU or statistical block and when combined this has the effect of delaying the implementation of an increase in catch.

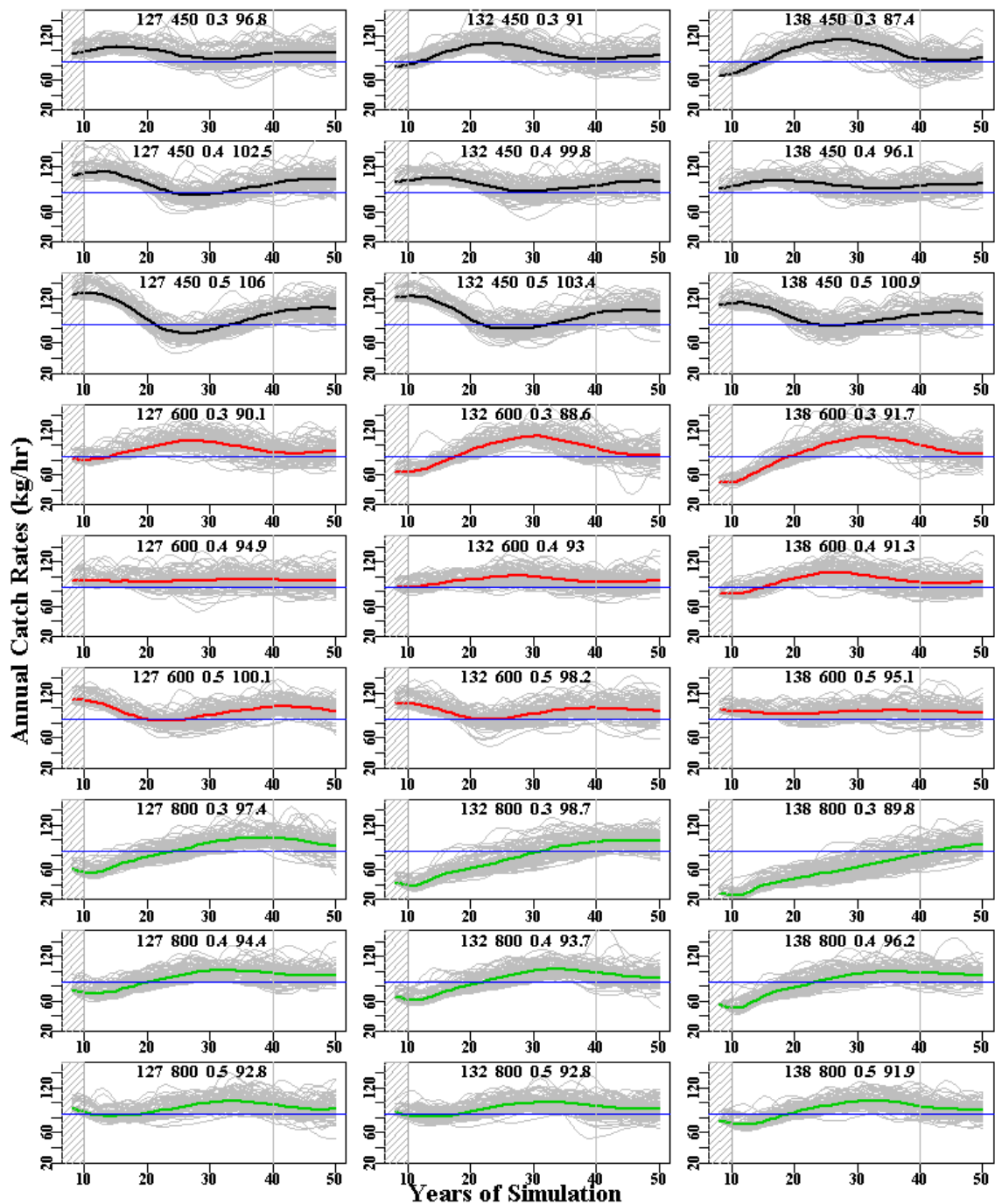


Figure 128. Envelope plot trajectories of the catch rates under each scenario where the TAC limit was implemented. Each graphs header includes the scenario but also the average catch rate over the final 10 years. The blue line in each case is the target and the grey vertical line identifies the year 40 from where the final average CPUE is estimated.

The bias is even more apparent when the catch rates for the final 10 years are plotted as box plots (**Figure 129**). The range of CPUE in the last 10 years is relatively wide, which reflects the variation across scenarios, however, in no case does the median quite meet the target CPUE and in some case sit is very far from it.

By delaying any increase in catch until the catch rates have risen above any particular target this ensures that the HCR generates biased management advice and the result is that the target is generally overshoot. First attempts at a solution to this issue found that the use of an alternative function ‘round’, which rounds each value to the closest integer did not suffer from the same problem. However, trials with an array of different target values indicated that the outcome was sensitive to the target selected and this appeared to be related to exactly how the values were rounded off and combined round appears to perform better with values like 85 and 95 kg/hr while trunk appears to operate reasonably well with targets that change in steps of 10 kg/hr. This issue of the HCR leading to a biased outcome will require further investigation. It might be thought that this HCR is at least biased in a conservative direction and so some may consider this acceptable. However, deliberately using a flawed solution instead of searching for a solution which enables one to aim for a specified target would reduce confidence in the clarity and transparency of the management advice so an effort should be made to improve on the HCR so that it performs according to specification.

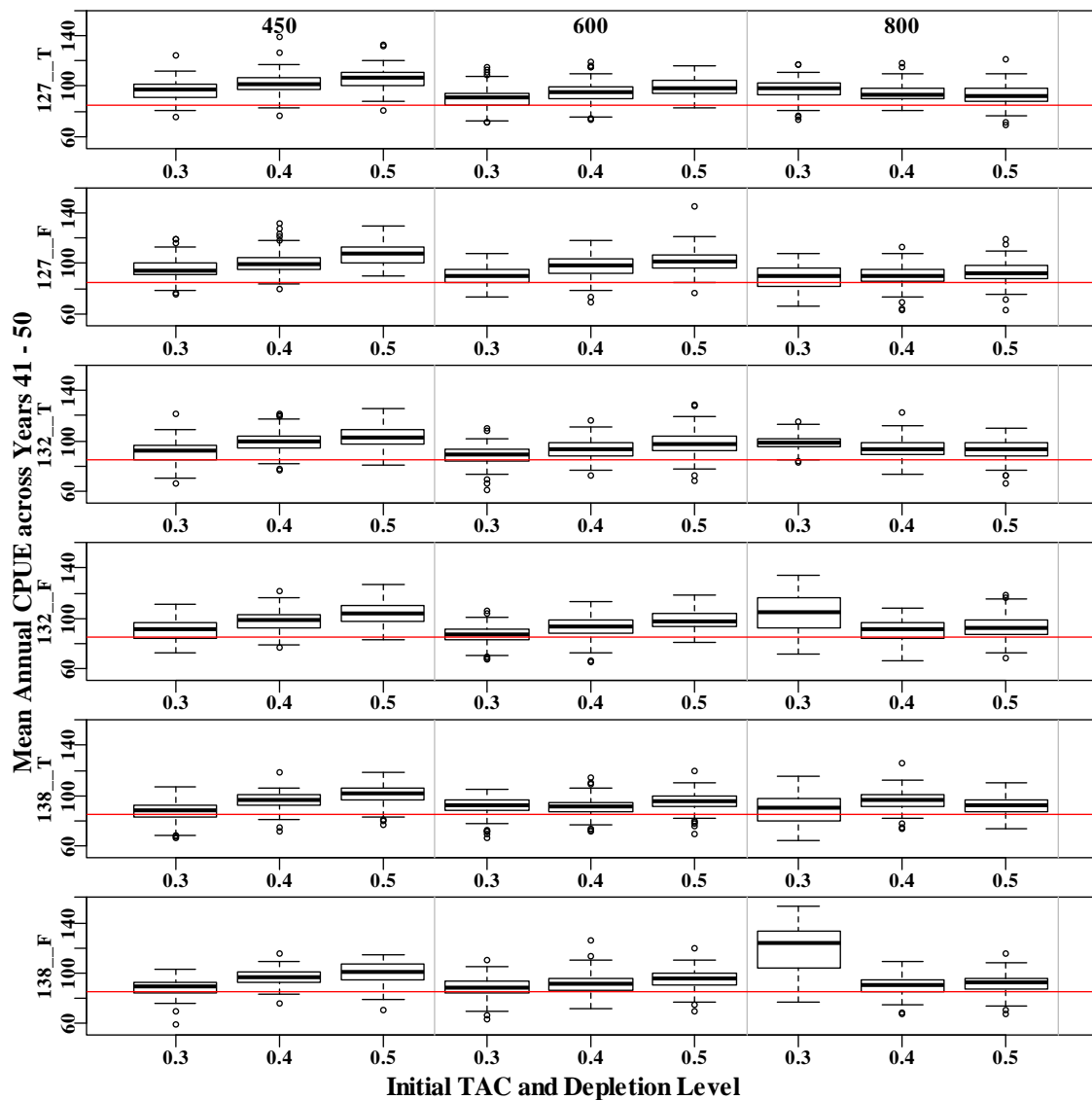


Figure 129. The mean annual catch rate (kg/hr) across the last ten years of the simulations for each of the 54 scenarios. The red line in each case is the target of 85 kg/hr.

21.5 Discussion

21.5.1 Gradient CPUE HCR

Despite the gradient HCR having some previous success in some other fisheries (Dichmont *et al*, 1999; Dichmont and Brown, 2010), the time lags in the abalone fisheries affected the performance of this harvest strategy to such a negative extent that it was essentially ineffective at providing useful management advice. Worse, it was unable to recover a depleted stock or optimize catches in a fishery starting from an un-depleted state. The time lags are too long relative to the dynamics of the stock and lead to management responses occurring too many years after any decline to have sufficient positive effects. The over- and under-compensation achieved with respect to catches simply took too long to feed through the dynamics of the stock to influence the CPUE that was the foundation of the HCR. These failures in the compensation led spawning biomass and catch rates to oscillate, in some cases with wide ranges between upper and lower levels.

21.5.1.1 *The Use of Mean Lengths*

There appeared to be little contrast in mean lengths of the catches in any of the scenarios so this performance measure may not have sufficient information to be of value in forming management advice. However, the mean length appeared to vary between different depletion levels so this PM is worth exploring further. Some means of including variation estimates as well as the mean value might capture more information about the status of the stock and be of more value in a multi-criterion decision analysis framework.

21.5.1.2 *The Initial Perturbation Option*

The option of introducing some contrast into the CPUE time series by perturbing the initial catch levels at the introduction of the HCR was considered. However, the perturbation explored (that of reducing initial catches by 25%) did not appear to be sufficient for the HCR to escape from its unintended behaviour when the stock was in a relatively depleted state; which is when this option would be required. Unfortunately, the initial perturbation option did little or nothing to improve the performance of the HCR.

21.5.1.3 *A Modified HCR Option*

It is possible that a way of using the CPUE gradient might be found and this would appear to require asymmetry to be introduced in the TAC response to the gradient. In this way it may contribute usefully in a multi-criterion decision analysis. Until those changes are explored further, however, it is recommended that this not be used in abalone fisheries. In particular, South Australia may wish to review its use of a symmetric scoring response to changes in CPUE, and that is despite noting that it doesn't use the gradient of CPUE but rather its occurrence within an acceptable range. Nevertheless, a symmetric response does not appear to dampen oscillations once they begin, nor does it act to stabilize the fishery.

21.5.2 Target CPUE HCR

The target CPUE HCR appeared to be much more effective at producing management advice that could permit the recovery of a depleted stock, and control the way a relatively un-fished fishery might develop.

21.5.2.1 *The Effect of LML on HCR Performance*

The assumption behind all the scenarios considered is that the simulated zone was similar to the eastern zone in Tasmania, which meant that the average BML of the zone was set at approximately 138mm. The productivity of the simulated zone in a deterministic sense was about 630 t, so it was not surprising that the 600 t initial TAC tended to be the most stable in all its outcomes across scenarios. Because an initial TAC of 800 t led to further depletion during the 10 year initiation period irrespective of the initial depletion level it was expected to produce much more variable outcomes across scenarios. However, what wasn't expected was that the 450 t initial TAC, which invariably allowed some rebuilding whatever the initial depletion level, also led to increased variability. It would appear that if you use the target CPUE HCR then oscillatory behaviour will arise if catches are too high or too low. As with the CPUE gradient HCR this appears to be a result of the time lags inherent in the relationship between catches, their impact on spawning biomass, and their consequent impact on subsequent recruitment. The time lags arise because of the time it takes for any new larvae to grow through the LML and become vulnerable to fishing, and hence contribute to future CPUE. The time lags lead to the HCR undercompensating for changes in the CPUE allowing CPUE and the related spawning biomass to over-shoot the target. By the time the HCR does respond to this over-shoot it then over-compensates and then under-shoots the target by continuing to increase catches beyond when it should if it was in fact aware of what the stock biomass was doing.

Given the history of abalone fisheries and the unspoken objective of maximizing catch in our fisheries it appears more likely that the various stocks are in a relatively depleted state than that they are in a relatively unfished state, but the 450 t and 50% initial depletion were included to provide contrast with the more likely 800 t and 30 or 40% depletion scenarios.

Focussing on the initial TAC of 800 t illustrates that the variation in catches, CPUE, and spawning biomass increases with the LML used and the initial state of depletion (meaning greater variation as initial depletion moved from 50% to 30%, which always became even worse under an initial TAC of 800 t. While the target CPUE HCR could manage a depleted stock back to a healthier position it could take a long time and also lead to relatively small catches if there is no TAC limit imposed.

21.5.3 The TAC Limit Option

Imposing a TAC limit certainly acts to maintain catches but it also acts to delay any recovery and this is exacerbated by increasing the LML. In many instances having a larger LML appears to make responses to change more variable and sometime more extreme.

It may appear that a higher LML of 138 mm relative to 127 mm acts in a disadvantageous manner even where the BML is 138mm, but this ignores the fact that if a stock is depleted down to the same level of catch rate rather than level of spawning biomass, then in all cases the spawning biomass in the 138mm LML scenario will be larger by at least 10%, which implies that it will be more resilient to environmental or disease events leading to mortality increasing in an episodic manner.

The use of a TAC limit can lead to large delays in recovery from a depleted state, the examples here suggested the difference could be as large as taking 2.5 times as long to

rebuild a stock to a productive state. Such a trade-off would still require discussion by policy makers, if the stock has the extra protection of a larger LML then the notion of a TAC limit may appear attractive for maintaining a market or making more divers economically viable within the fishery. Unfortunately, there is no simple way of determining at what level such a TAC limit should be set. There is a gradient where all such a limit does is delay recovery and the higher the limit the longer the delay. At the end of that gradient, however, if the TAC limit is set too high, and what is too high remains unknown in the real world, then the delay in rebuilding passes a threshold and the stock either stays stable and low or declines still further, albeit slowly. A comparison with the recent situation in the most productive region in New South Wales was used to suggest that the long period of relatively low catch rates (an average of ~ 23 kg/hr from 1988 – 2004) was due to catches being held too high (an average of ~77 t from 1988 – 2004). But when these catches were almost halved down to an average of 40 t per annum the catch rates appear to have recovered.

Similarly, if a TAC limit is installed the risk of maintaining a stock in a depressed state, even with a relatively large LML, should be considered carefully.

21.5.4 Issues with the Target CPUE HCR

The use of the CPUE target HCR appears to be a viable option, especially in the context of the multi-criterion decision analysis framework, which permits other fishery performance measures to be included in the decision making. However, the details of this HCR are not yet operating as well as it might. The use of the current proposed algorithm, which truncates the various scores for each Statistical Assessment Unit assessed down to the lowest integer, has the effect of biasing the response low. This means that the HCR is less responsive to changes in CPUE than it could or should be. The overall effect of this is that the HCR tends to lead to attaining a target that is somewhat above the selected target. The truncation of the scores means that even though the observed CPUE may be higher than the target, the scores deriving from the observations give the appearance that the fishery is at the target rather than above it.

Because this flaw is known it does not rule out the use of this HCR, however, further work is desirable so that it performs more effectively at achieving the selected target.

22 References

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