Stock Assessment and Management Strategy Evaluation for Sub-Antarctic Fisheries

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1 NON-TECHNICAL SUMMARY

2000/109	**Stock Assessment And Management Strategy Evaluation For Sub-Antarctic Fisheries**

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OBJECTIVES:

1. To provide the SAFAG with updated information on the current status of Patagonian toothfish around Macquarie Island, including development of extended stock assessment models that link the current tag-based models to age-structured and spatially structured population models, and to commercial catch data.
2. To develop long-term management strategies for the Macquarie Island Patagonian toothfish fishery.
3. To participate in the stock assessments for Heard Island and McDonald Islands Patagonian toothfish and icefish conducted through CCAMLR, and assist in providing effective communication between the CCAMLR and AFMA assessment processes.

NON TECHNICAL SUMMARY:

OUTCOMES ACHIEVED

The assessment of stock status of Patagonian toothfish at Macquarie Island continues to be based on the methodologies presented in this report. The assessment outputs are a critical input to the management and TAC setting process for this fishery. The results from the Management Strategy Evaluation are being used by SAFAG, industry and management to help manage the fishery in accordance with agreed sustainability objectives. The results of the project have increased fishers' and managers' awareness of the utility of setting and evaluating appropriate management strategies for fisheries. In addition, the application of assessment methodologies developed under this project to CCAMLR fisheries have complemented and enhanced existing CCAMLR assessment procedures.

The Australian trawl fishery for Patagonian toothfish *Dissostichus eleginoides* began in November 1994 in the waters surrounding Macquarie Island. Only a single vessel, the *Austral Leader*, has been licensed to fish in the area. All fishing has taken place in the spring and summer between the months of October and March, except during 2000, when a fishing voyage was undertaken in June. During the first two seasons, the Aurora Trough ground was established and catches and catch rates increased as knowledge of the grounds and fishing techniques improved. A second fishing ground, the Northern Valleys, was discovered in the

1996/97 season. Since then, both fishing grounds have been targeted. In the 1995/96 season a tagging program was established by the Australian Antarctic Division to assist stock assessments and monitor fish movement. By the 2000/2001 fishing seasons, over 4700 Patagonian toothfish had been tagged, of which over 500 were recaptured.

The stock assessment models provide estimates of the historical abundance of toothfish accessible (or available) to the fishery. Available abundance is quite distinct from total abundance and spawning abundance, which may be substantially more extensive than available abundance both spatially and in total magnitude. The assessment model based on the tagging experiment estimated that pre-tagging available biomass in Aurora Trough was between 3 and 5 thousand tonnes and in the Northern Valleys region between 30 and 45 thousand tonnes. Confidence limits on estimates of pre-tagging abundance for the Northern Valleys are very broad due to the recapture of only three tagged fish in the first season. This region shows a dramatic decline in available abundance between its first (1996/97) and second (1997/98) seasons. This reduction in available biomass is greater than can be explained by the fishery catches even in the absence of recruitment. For Aurora Trough, estimated available abundance declined over the first three seasons of tagging (to 1997/98), but has since shown an increasing trend. This increase corresponds well to the beginning of catch restrictions for commercial fishing in the Aurora Trough region (a 40t research quota only was allowed). Relative abundance trends from fishery catch and effort analyses also suggested that a decline in available abundance had occurred, with a small increase in relative abundance in season 2000/01. However, the level of the initial decline is uncertain, as the indices are sensitive to the data (particularly the poor contrast between model factors) and model structure.

Hypotheses being explored to explain the dynamics of the Macquarie Island region, and in particular the Northern Valleys, include (i) that the population is composed of a small resident population, but that most fish are part of a broader transient population, and (ii) that the fish have remained in the area but their availability fluctuates greatly over space and time (due to dispersion in the water column, moving to untrawlable ground or escaping detection). This uncertainty indicates that an understanding of the fishes' behaviour and movement is important.

The tag-recapture model was also applied to one of the main grounds in the Heard Island and McDonald Islands toothfish fishery. The HIMI toothfish fishery began in April 1997, with two vessels licensed to fish the region, the *Austral Leader* and *Southern Champion*. Tagging commenced in May 1998 and around 1000 fish have been tagged each season since then. The estimates of available numbers of fish in this ground declined steadily during the course of the experiment and are now at just under 50% of the value when the tagging program began. However, illegal fishing is known to occur in the region and its influence on estimates of available biomass is difficult to determine. Further development of the tag-recapture models may shed some light on the degree of illegal fishing in the area. Nevertheless, the ability to estimate the available abundance using tagging data complements and further enhances the assessment methodologies (such as the Generalised Yield Model) already developed to manage the fishery.

The uncertainties surrounding the dynamics of Macquarie Island toothfish create a challenge for management to adopt strategies that recognise these uncertainties, and yet are capable of achieving the management objectives of sustainable fishery development. Management objectives for the Macquarie Island fishery were discussed at workshops and sub-Antarctic fisheries assessment group (SAFAG) meetings attended by managers, industry and scientific representatives. To achieve the specified management objectives (conservation and utilization), management strategies are formulated that provide a set of pre-agreed rules for selecting management actions. The method used to evaluate the performance of alternative management strategies under various resource assumptions that encompass the system's uncertainties is called Management Strategy Evaluation (MSE).

The basic method for evaluating performance uses repeated stochastic simulations. The method involves the simulation of the fishery from its beginning to a pre-determined future year (2050 in this instance). An 'operating model' is used to simulate the 'true' dynamics of the toothfish population and the fishery. Once a set of assumptions about the dynamics of the resource (a scenario) and a management strategy have been chosen for evaluation, the biology and fishery dynamics are simulated over historical and future years. For each projected year the management strategy prescribes the form of annual assessment (and even whether one is undertaken), the sampling effort, and the annual catch limit. The annual catch limit may be a function of the assessed current or past status of the stock, but may also be unrelated to current knowledge of the stock (e.g. if no TAC is set). Performance statistics are combined over all simulations and tabulated to provide a summary of the performance of the particular management strategy for a given scenario.

Three underlying resource scenarios were considered: (i) a single stock model, (ii) a single stock model with periods of zero recruitment (iii) a two stocks model. The first resource model assumes that the harvested population is reproductively isolated from external sources of immigration. The second resource model is similar to the first except that the population occasionally experiences recruitment failure occurring over a number of consecutive years. This resource model has been included to model the apparent poor observed recruitment of young fish in this fishery. The third resource model assumes that an external transient population of toothfish exists that occasionally moves into the region occupied by a resident population. The first two models are believed to represent the situation occurring in Aurora Trough, whereas the third model attempts to account for the dramatic estimated changes in fishable abundance in the Northern Valleys region. Clearly these resource models vary considerably in their assumptions and one of the aims of management strategy evaluation is to provide management strategies that are robust (in terms of performance) across all resource models and other key uncertainties.

The results suggested that fishing operations at Macquarie Island have had a smaller impact on the resource than one might expect. However, this result is conditional on several key factors. These are that (a) selection applies to a narrow range of ages, (b) spawning fish contribute to local recruitment only, (c) economic constraints lead operators to cease fishing and depart the region in poor catch conditions, (d) effort is limited, and (e) assessment estimates of key biological parameters are reasonable. If any of these factors are false, then results show that the impact on the stock can be substantial.

An analogous management strategy to that used in the current fishery outperformed others considered. This strategy assigns as the TAC a fraction of the estimated available biomass from an annual assessment and allows an increase in TAC if catch conditions are favourable. However, it did falter (as did the others) in one key circumstance. Namely, resident stock spawning biomass may reduce to unacceptable levels if effort is markedly increased, there are two stocks present and no management intervention occurs when stock signals would indicate a reduction in spawning biomass. Under this extreme scenario, fishing continues through periods of poor catch when the operators would normally have chosen to leave. The presence of transient fish leads to increased estimates of annual available biomass and hence TACs. The increased effort allows more of the TAC to be caught and subsequently more of the resident population is taken as part of the annual catch. This problem arises as the current assessment model assumes that all catch and tagging comes from a single population. The ensuing available biomass, which in this case is a combination of stocks, is used to set the following year's TAC. However, MSE results showed that a TAC decision rule based on the resident stock alone could meet conservation criteria for the population. However, this initial result depended on perfect knowledge of resident available biomass, which is normally estimated by a stock assessment model. As such, stock assessment models of two populations, resident and transient, were developed and initial testing proved promising.

Stemming from the MSE work, an application of Bayesian Belief Networks (BBN) was developed to explore the relative belief of alternative hypotheses about stock structure and movement of the toothfish population and its influence on management. In the network, all relevant information, including expert opinions, field observations, and model outputs, can be incorporated (and updated), with greater weight given to information that has less uncertainty and variance. For this project, the BBN was used to explore alternative hypotheses of stock structure in the two main fishing grounds of Macquarie Island. Fishery and research data from tagging, genetics, fatty acid analysis, stock assessments, and fishery catches, as well as other biological information, were included in the BBN models. Results showed how the predictive nature of the model changes according to the information received. One of the many results of this study showed that information gathered during the project (i.e. that there is no genetic difference between the grounds, there was a fatty acid difference, and no adult migration between grounds) increased the likelihood that transient fish exist. The models also have the ability to look at the benefits of biological and monitoring information through their impact on belief in stock hypotheses (for example, how would our belief in the biological hypotheses differ if the tagging or fatty acid programs had never occurred?).

KEYWORDS: Patagonian toothfish, Stock assessment, Macquarie Island, Heard Island and McDonald Islands, Management strategy, Bayesian Belief Networks

2. BACKGROUND

Geoff Tuck

Bottom-set longline and trawl fisheries for the Patagonian toothfish (*Dissostichus eleginoides*) developed in the waters of several of the Southern Ocean's sub-Antarctic islands during the late 1980s and early 1990s. More recently, trawl fisheries for toothfish were established within Australian Commonwealth waters around Heard Island and McDonald Islands (HIMI) and Macquarie Island.

The Patagonian toothfish is a large, long-lived, bottom-dwelling species inhabiting the continental shelf waters of sub-Antarctic islands, oceanic ridges and the southern South American continent. Patagonian toothfish is a highly prized table fish with significant imports to Japanese, North American and European Union markets.

Toothfish have been known to grow to over 2m in length and may live to more than 50 years of age. They inhabit depths from approximately 300m to 2000m, with juveniles generally found in shallower water. They feed on small fish and squid in the mid-water and various fish and crustaceans on the bottom. Toothfish are believed to reach sexual maturity at around 10 years of age, and possibly older for Macquarie Island fish (Constable et al., 2001; Goldsworthy et al., 2001).

Toothfish lack swim-bladders and so often reach the surface in good condition even though they may have been trawled from depths of 400m and more. This has allowed an extensive tagging program to develop at both Macquarie Island and HIMI. Tagging studies have increased knowledge of the species movement, growth and available abundance (Williams et al., 2002; Tuck et al., 2003). Estimates of the available abundance of toothfish from the tagging program are described in more detail within Section 5.

Macquarie Island

Macquarie Island lies some 1500km to the southeast of Tasmania (Figure 2.1). The fishery off Macquarie Island began in November of 1994 with only one vessel, the *Austral Leader*, licensed to fish the Macquarie Island toothfish stock. Two major fishing grounds have been discovered: Aurora Trough and the Northern Valleys region. A tagging experiment began in 1995/96 within Aurora Trough and the following season within the Northern Valleys region. Between 1995/96 and 2000/01, over 4700 tags were deployed and over 500 tagged fish were recaptured. Only two tagged fish were recaptured having migrated between the two main fishing grounds during this period.

A Total Allowable Catch (TAC) for the fishery was first introduced in the 1996/97 fishing season. The TAC for the 1996/97 fishing season was based on the catches of the first two fishing seasons and the tagging experiment in the 1995/96 fishing season. The setting of TACs after the 1996/97 fishing season was then based on results from the stock assessment models. For the Aurora Trough region, TACs were 750 and 200 tonnes for the 1996/97 and 1997/98 fishing seasons respectively, and were zero after the 1997/98 fishing season (but with a 40 tonne research TAC for continuing the tagging experiment and monitoring). For the Northern Valleys region, TACs were 1000, 1500, 600, and 540 tonnes for each fishing season from 1996/97 to 1999/2000 respectively. However, the TACs for the last two fishing seasons (1998/99 and 1999/2000) were allowed to increase within the fishing season if the catch rates exceeded $10t/km^2$ over three consecutive fishing days (see Section 5). If this catch rate dropped below the trigger level, then the TAC fell to the lower TAC. If the lower TAC had been reached then fishing ceased.

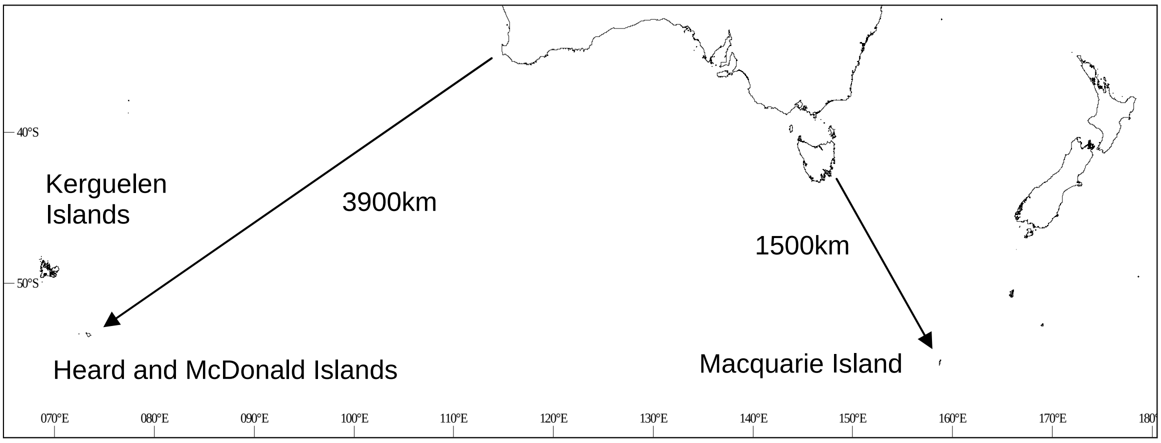

Figure 2.1. The location of Macquarie Island (54°30'S, 158°57'E) and Heard Island and McDonald Islands (53°06'S, 73°30'E) relative to New Zealand and Australia.

The assessment of the Macquarie Island Patagonian toothfish stock is based on the tag-recapture model developed by de la Mare and Williams (1997), and modifications described in Tuck et al. (2003), and indices of relative abundance determined from a standardisation of catch and effort data. The tag-recapture model uses population and tag accumulation models to account for tags being released at various times throughout the season and the effects of removals by fishing. The model is able to estimate pre-tagging available abundance and annual recruitment (net change in available abundance between seasons) of Patagonian toothfish within the major fishing grounds of Macquarie Island.

Catch rates, standardised in order to produce relative indices of abundance, are also estimated for the Macquarie Island fishery (Section 6). Standard statistical methods that use General Linear Model methodologies are employed to produce the indices, which are assumed to be proportional to the density of the sampled population (McCullagh and Nelder, 1989). The statistical standardisation attempts to account for changes in the fishery (e.g. gear changes, skippers, fishing depth) and the environment (e.g. sea surface temperature, moon phase) that may bias unstandardised catch per unit effort values.

Table 2.1. The Total Allowable Catch (TAC) for the Macquarie Island Patagonian toothfish fishery. The total quota for the Macquarie Island fishery could not include more than the prescribed amount from the Aurora Trough region. * Measures were available to allow an increase in the TAC if the average catch per unit effort exceeded a specified amount.

	TAC (tonnes)	
Administrative Period	Aurora Trough	Total Macquarie Region
1 Sept 1996 – 31 Aug 1997	750	1000
1 Sept 1997 – 31 Dec 1998	200	1500
1 Jan 1999 – 31 Dec 1999	40 (research)	600*
1 Jan 2000 – 31 Dec 2000	40 (research)	510*
1 Jan 2001 – 31 Dec 2001	40 (research)	420*

Assessments of the population of toothfish at Macquarie Island have indicated that the population has undergone major changes in fishable abundance since the fishery began in the summer of 1994/95 (Sections 5 and 6). The spatial and temporal dynamics of the population remains largely uncertain. For example, it is unclear whether the population at Macquarie Island is a single well-mixed stock, or is composed of two or more local populations, or whether transient fish having a more cosmopolitan habit occasionally join the resident fish. To a large extent, several other key components of the fishes' biology also remain uncertain. The challenge for managers is to establish methods for managing the harvested populations that recognise these uncertainties, and which satisfy, to a reasonable degree, various, often conflicting, management objectives

A management strategy is a set of pre-agreed rules for selecting management actions, designed to achieve specified management objectives. In the current context, the components of a management strategy are the sampling program, the assessment and a decision rule that translates the data from the sampling program and information from the assessment into a TAC. The method used to evaluate these strategies is called Management Strategy Evaluation (MSE). MSE does not attempt to find an 'optimal' strategy, but rather explicitly outline the trade-offs inherent in managing stocks with potentially competing objectives. The aim of MSE is to consider alternative management strategies and examine their performance against a range of management objectives under various assumptions that encompass the system's uncertainties.

Section 7 describes the various ecological scenarios and management strategies considered for the Macquarie Island fishery. The MSE analyses are followed by an application of Bayesian Belief Networks to the Macquarie Island fishery, which stems from the MSE work. These methods explore the relative belief of alternative hypotheses about stock structure and movement of the toothfish population. The BBN models are based on causal networks where alternative hypotheses are linked to observable variables. In the network, all relevant information, including expert opinions, field observations, and model outputs, can be incorporated, with greater weight given to information that has less uncertainty and variance. The BBN can then be updated with new information or belief to estimate posterior probabilities of the alternative hypotheses. For this project, the BBN was used to explore alternative hypotheses of stock structure, movement, and locality of toothfish in the two main fishing grounds of Macquarie Island. Fishery and research data from tagging, genetics, fatty acid analysis, stock assessments, and fishery catches, as well as other biological information, was included in the BBN models.

Heard Island and McDonald Islands

While not the main focus of the project, the tag-recapture assessment models developed for the Macquarie Island fishery were applied to Heard Island and McDonald Islands toothfish to complement the assessments conducted through CCAMLR (Constable et al., 2001). Heard Island and McDonald Islands are located approximately 3900km to the south west of Australia in the southern Indian Ocean (Figure 2.1). Along with the Kerguelen Islands some 440km to the north-west, they are the only land masses that belong to the Kerguelen Plateau.

The Australian Exclusive Economic Zone (EEZ) of Heard Island and McDonald Islands lies within the CCAMLR Convention Area. The fishery falls within CCAMLR statistical division 58.5.2. Details of the management arrangements for the Heard Island and McDonald Islands fishery can be found in the Strategic Assessment Report (AFMA 2002). The trawl fishery began in April 1997, with tagging commencing in May 1998. Around 1000 fish have been tagged each season since then. There are three main fishing regions, denoted Fishing Grounds A, B and C. Assessment of the available abundance of toothfish from the tagging data was only attempted for Ground B because of the relatively low number or sporadic nature of recaptures in the other grounds (Williams et al. 2002).

3. NEED

Geoff Tuck

The Patagonian toothfish fishery is expanding worldwide and may play a pivotal role in the development of an Australian fishing industry in the Southern Ocean. Given the conservation values of Macquarie Island and Heard Island and McDonald Islands, it is expected that fishery operations will be closely scrutinised. These fisheries provide an opportunity to illustrate that fishery development can be achieved while protecting conservation values – a demonstration of Ecologically Sustainable Development (ESD).

Given AFMA's need to satisfy its ESD objectives, there is an ongoing need to continue the stock assessment and management strategy evaluation process for these fisheries. The Sub-Antarctic Fisheries 5-Year Strategic Research Plan highlights the need to "develop and review appropriate stock assessment models for toothfish" and to further the management strategy evaluation approach for Macquarie Island. This project provides estimates of current stock status, furthers our understanding of the dynamics of the populations, and evaluates alternative management strategies for the Macquarie Island fishery. The results of this project as presented to the Sub-Antarctic Fisheries Assessment Group (SAFAG) will facilitate the setting of an annual TAC for Macquarie Island Patagonian toothfish and provide valuable input to the assessment process for the Heard Island and McDonald Island fishery.

While primarily focused on assessing the toothfish fishery at Macquarie Island, the project has included participation in the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) assessments of the toothfish and icefish fisheries at Heard Island and McDonald Islands. This participation addresses the need to improve coordination, communication and mutual support between the international and domestic assessment processes applied to Australia's sub-Antarctic fisheries.

4. OBJECTIVES

Geoff Tuck

1. To provide the SAFAG with updated information on the current status of Patagonian toothfish around Macquarie Island, including development of extended stock assessment models that link the current tag-based models to age-structured and spatially structured population models, and to commercial catch data.

2. To develop long-term management strategies for the Macquarie Island Patagonian toothfish fishery.

3. To participate in the stock assessments for Heard Island and McDonald Islands Patagonian toothfish and icefish conducted through CCAMLR, and assist in providing effective communication between the CCAMLR and AFMA assessment processes.

5. THE TAG-RECAPTURE ASSESSMENT

Geoff Tuck, Dick Williams, Tony Smith, Xi He, Andrew Constable and Tim Lamb

5.1 Introduction

The abundance of a population is often estimated by the joint analysis of tag-return and catch data by the Petersen approach (Petersen (1896), as cited in Ricker (1975)) and extensions that allow estimates of recruitment (Fischler, 1965; Jolly, 1965; Seber, 1965; Manly, 2002). Semi-parametric models, where some model attributes are specified parametrically and others non-parametrically, have been used to analyse daily release and recapture tagging data when it is not feasible to specify fishing mortality rates as daily parameters (Hearn et al., 1987; Leigh, 1988; Barndorff-Nielsen et al., 1989; Xiao and McShane, 2000). As these models only analyse tag-return data and do not include catch information, they can only estimate mortalities, not population abundance.

Stock assessment models, incorporating tag-return data, have been developed that estimate population sizes (e.g. Patterson, 1999). However, these models pool data into annual cells and do not account for delayed mixing or the recapture of newly tagged fish during the tagging operation. The model used to assess the Macquarie Island Patagonian toothfish stock is a modification of the de la Mare and Williams (1997) model that unifies the semi-parametric approach with the Petersen method. It gives an estimate of the pre-tagging abundance of Patagonian toothfish accessible to the fishery and the net annual recruitment. We assume that the population of interest is distinct (e.g. Aurora Trough and the Northern Valleys regions of Macquarie Island and Ground B of Heard Island and McDonald Islands). We also assume that tag recaptures are conditional on catches and the estimated abundance of tagged fish, and are Poisson distributed.

It is important to clearly distinguish between several inter-related and often confused terms. The stock assessment method proposed here is only able to estimate *available* abundance (also referred to as *fishable* abundance). Available abundance is quite distinct from *total* abundance and *spawning* abundance, which may be substantially more extensive both spatially and in total magnitude. Total abundance includes every individual (juvenile and adult) within a reproductively distinct population (whether available to the gear or not). Spawning abundance is the abundance of those fish of the population that are sexually mature.

There are two factors which influence the available abundance (i) *gear selectivity* (the probability of capture on encounter) and (ii) *availability* (the probability of encounter by the fishing gear (Marr 1951)). While gear type will influence gear selectivity, oceanographic conditions and population movements can strongly influence the availability of fish. These two factors are generally functions of age or length, and can vary within and between seasons. In this report the combined function of gear selectivity and intra-annual availability at age is called the *vulnerability* function. The vulnerability function for Macquarie Island toothfish has been estimated by Constable et al. (2001) and is discussed later in this section.

5.2 Methods

Tag releases and recaptures occur throughout the fishing season at Macquarie Island and Heard Island and McDonald Islands. We could accumulate each of the catches, releases and recaptures over the fishing season and naively estimate the available abundance by the Petersen method. However, the available abundance could be seriously overestimated as this approach disregards the expectation that a fish tagged early in a fishing season has a higher chance of being recaptured during the season than a fish tagged later in the season. We use a semi-parametric model to account for daily catches, releases and recaptures and assume that there is neither recruitment nor emigration during the fishing season. Net recruitment to the available stock occurs between seasons and can be estimated when more than one year of tag returns are available.

The probability of recapturing a tagged fish depends on the size of the catch and the size of the fishable stock. The total expected number of tag returns from a single catch will then depend on the probability of recapture and the number of tagged fish in the water (Petersen's equation[1]). This expectation will vary with time as tagged fish are released and recaptured, and as the size of the population changes with catches, natural mortality and recruitment.

The population models of the assessment include dynamics of tagged and untagged fish, allowing for natural and fishing mortality, net recruitment, growth and daily releases and recaptures. It is assumed that there is neither recruitment nor emigration during the fishing season. The parameters estimated by the model are the pre-tagging available abundance N_0 and the annual net recruitment, R_y (defined as the net change in available abundance between seasons). Daily estimates of available biomass can also be tracked.

Between the 1995/96 and 2000/2001 fishing seasons, over 4700 Patagonian toothfish were tagged in the waters around Macquarie Island, of which over 500 have been recaptured. Likewise, in the Heard Island and McDonald Islands fishery around 5200 fish have been tagged with 738 recaptures. Although toothfish are trawled from more than 400m depth, tagging is straightforward because toothfish lack swim-bladders and reach the surface in good condition. Lively fish are selected from the pounds and placed in a holding tank of circulating seawater. A length range of fish is chosen that approximates the length range in the catch. The fish are tagged with electronic tags (TIRIS radio frequency identification transponders) and a visible plastic tag (double tagged since 1997/98). The fish are then replaced in the holding tank for about 30 minutes after tagging to check on vital signs and then released through a scupper in the factory. In the 1996/97 season, an electronic tag detector was installed on the vessel. Unfortunately, the electronic detector was initially not fully effective, so less than 100% of the electronic tags were detected.

Tag-recapture experiments rely on the tags being discovered and reported when the fish are captured. This may not occur if tags are lost from the fish, or if tagged fish are not detected. From the recapture of multiple tagged fish in this fishery, estimates of tag loss rates indicate that the probability of losing both tags is negligible. Likewise, as many individual fish have been recaptured several times, tagging mortality was assumed to be zero.

The non-detection of tagged Patagonian toothfish has been a problem, especially with the electronic tags. The detection of visible tags also relies upon the vigilance of the crew and

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[1] Provided tagged fish have mixed with the un-tagged population, Petersen's equation states that the proportion of marked fish in the population, m/N, should equal the proportion of recaptured fish in the catch, r/C. Rearranging this equation for recaptures yields, $r = Cm/N$.

observers. If it is assumed that the probability of detecting the two types of tags is independent, the total number of recaptured tagged fish in a season, r_s, can be estimated by the moment estimator[2]:

$$r_s = n_v n_e / n_{ve}, \tag{5.1}$$

where n_v and n_e are the number of tagged fish detected either visually or electronically, and n_{ve} is the number detected by both methods.

5.2.1 Model formulation

a) The Length-Independent Selection Model (LIS)

Let N_t be the number of available fish in the population on day t, and C_t the number of fish caught on day t, then:

$$N_t = (N_{t-1} - C_{t-1})S, \tag{5.2}$$

where $S = exp(-M/365)$ and M is the annual natural mortality rate, which is assumed constant for all time periods. This is the daily version of the catch equation used in cohort analysis (Pope, 1972).

The number of available fish prior to the tagging experiment is given by N_0 and the reference day is taken as 1 July. One year later, R_y recruits are added to the population. The between-season 'net recruitment' is a measure of the net change in available abundance between seasons. Hence, recruitment (in the usual sense) of young fish is not explicitly modelled, but is aggregated with availability and movement effects.

Again, using the cohort catch equation, the number of tagged fish in the population on day t is:

$$m_t = (m_{t-1} - r^*_{t-1})S + p_t, \tag{5.3}$$

where p_t is the number of tagged fish released on day t, and $m_{t_1} = p_{t_1}$ where $t=t_1$ is the first day of tagging. The total number of recaptured fish r^*_t on day t is given by:

$$r^*_t = r_t / \omega_t, \tag{5.4}$$

[2]This equation derives from the probability of detecting tags by both methods being the product of the probabilities of detecting each type of tag independently. Replacing probabilities with their sample frequencies and rearranging for the unknown total recaptures r_s yields equation (5.1).

where r_t is the observed number of recaptures and ω_t is the detection rate for day t. The number of observed recaptures on day t is assumed to follow a binomial distribution, $r_t \sim B(\beta_t, C_t)$, with mean μ_t defined by:

$$\mu_t = E[r_t] = \omega_t \frac{m_t}{N_t} C_t = \beta_t C_t , \tag{5.5}$$

where $\beta_t = \omega_t m_t / N_t$ for day t. As there are large catches (or samples) and m_t / N_t is small, the Poisson distribution Po approximates the binomial distribution, and thus $r_t \sim Po(\mu_t)$. It is assumed that recaptures are random and not clumped. We validated this assumption with Greene's (1993) algorithm.

Maximum likelihood estimates of pre-tagging available abundance, N_0, and net recruitment in year y, R_y, are then found by maximising the log-likelihood function given by:

$$L(r; N_0, R_y) = \sum_{t:\mu_t \neq 0} (r_t \ln(\mu_t) - \mu_t). \tag{5.6}$$

b) The Length-Dependent Selection Model (LDS)

The Patagonian toothfish caught in the trawl fisheries at Heard, McDonald and Macquarie Islands are predominantly between 450 and 1000mm long (Constable et al., 2001). This observed length-frequency is related to both gear selectivity and fish availability. Evidence from deep-set longline toothfish fisheries at other sub-Antarctic Islands where large fish are caught in greater numbers, and the infrequent but evident catch of large fish at Macquarie Island, suggests that toothfish availability decreases with increasing length. This is likely to arise as a result of smaller fish being segregated from adult fish, with the adults tending to be found in deeper water (SC-CAMLR, 1998). The LIS model assumes that all tagged fish remain available to the gear regardless of age or length. As such it is possible that the model assumes there are more tagged fish available than there are in reality, which would lead to an over-estimation of available abundance. This is because Petersen's equation (and the LIS model) assumes that abundance is proportional to the number of tagged fish in the population. Hence, for the same catch and number of recaptures, the population estimate will be biased upward if the number of tagged fish available is, in fact, lower than anticipated.

The LDS model assumes that the likelihood of recapturing a released fish is a function of vulnerability. Initially, every released fish is aged using its recorded release length and the von Bertalanffy growth curve, given by:

$$a = t_0 - \ln(1 - l / L_\infty) / k, \tag{5.7}$$

where l is the length, L_∞ is the asymptotic length of a fish, k is a growth parameter and t_0 is the age at which a hypothetical fish is of length 0cm (Quinn and Deriso, 1999). The modelled-tagged fish then grow each day according to the von Bertalanffy growth curve with length a function of age.

The expected number of surviving available tagged fish[3] on day t is,

$$m_t = \sum_j s(l_t^j)\exp(-(t - t_r^j)M / 365) \tag{5.8}$$

where j represents all releases (recaptured or not) prior to day t (akin to Hearn et al. (1987)). Times t_r^j and t_c^i are the day of release and recapture for a tagged fish, where $t_r^j < t$. The function $s(l_t^j)$ is the vulnerability of tagged fish j, which has length l_t^j on day t. The exponent term gives the probability of natural survival of tagged fish j from the day of its release t_r^j to the current day t. Equation (5.8) replaces equation (5.3) in the description for the LIS Model.

5.2.2 Sensitivity to mixing

As the tagged fish are released in a group, they possibly do not mix with the un-tagged population for some days. This can bias estimates of abundance. For example, if tagged fish remain near the vessel and are recaptured shortly after release, then the assessment model is likely to under-estimate abundance. Likewise, if tagged fish (or the vessel) move to another region, tagged fish will be under-represented in the catch and estimates of abundance will be biased upward.

We follow the simple procedure used by Hearn (1986) to allow a period of time for tagged fish to mix with the general population. Note that the equivalent procedure for parametric models is more complex, unless the mixing period is one year (Hoenig et al., 1998). This is an advantage with using a semi-parametric model over the corresponding parametric one.

Let:

δ = the minimum number of days for tagged fish to fully mix with the un-tagged population.

To explore the sensitivity of the model to δ, tagged fish recaptured within δ days are removed from the analysis, i.e. both the release and recapture events are removed from the input data. As released fish cannot contribute to the expectation of recapture within δ days of their release, their inclusion to the formulation for the number of tagged fish in the water (equations (5.3) and (5.8)) can not occur until δ days after their release. At this point they have experience δ days of natural mortality.

5.2.3 Confidence Limits

The likelihood profile method (as defined by Hilborn and Walters, 1995) is used to calculate a 95% confidence set around a point estimate of a parameter, ρ. The confidence limits defining the boundary of the confidence set are defined as those values of ρ_{sub} that satisfy:

$$L(r; \rho = \rho_{sub}) - L(r; \rho = \rho_{opt}) = \chi^2_{1,1-\alpha} / 2, \tag{5.9}$$

---

[3] Note that, subsequent to Tuck and Williams (2001), this formulation now includes an adjustment for tagged fish not detected, in a similar manner to equation (5.4). See Tuck et al. (2003) and Appendix C.

where $L(r;\ \rho = \rho_{opt})$ is the negative log-likelihood corresponding to the maximum likelihood estimate (giving ρ_{opt} as the point estimate), and $L(r;\ \rho = \rho_{sub})$ is the lowest negative log-likelihood when ρ is set to the sub-optimal value ρ_{sub}. The value of the chi-squared distribution with one degree of freedom at confidence level 1-α is given by $\chi^2_{1,1-\alpha}$. For $\alpha = 0.05$, the right-hand side is equal to 1.92.

5.3 Macquarie Island

5.3.1 Results

Separate analyses are conducted on the Aurora Trough and Northern Valleys region as part of the annual review and assessment process. The grounds are treated in isolation for management purposes. This assumption is supported by a preliminary genetic study (Reilly et al., 2001) and the apparent lack of movement between regions suggested by the tagging experiment (Williams and Lamb, 2001). Only 2 tagged fish have been recaptured having transferred between Aurora Trough and the Northern Valleys. While the applications described in this report focus only on these regions, numerous other regions surrounding Macquarie Island have been explored (including the release of tagged fish from these regions). These areas are not analysed here.

Table 5.1 shows the number of tags detected by each method by season for the Aurora Trough and Northern Valleys fishing grounds. For example, in the 1996/97 season $n_v = 33 + 36 = 69$, $n_e = 11 + 36 = 47$ and $n_{ve} = 36$, giving an estimate of $r_s = 90.08$. As a total of 80 fish were detected, the estimated detection rate over days t when both visual detection and electronic detection were operational is $\omega_t^{ve} = 0.89$. A detection rate based only on the visual detection of tags, ω_t^{v}, can also be calculated using this method. Noting that ω_t^{ve} does not apply outside of the period when both detection mechanisms are possible, we have used the visual detection rate for periods when the TIRIS detector was not functional. The estimated visual detection rate for 1996/97 is used for season 1995/96, as the electronic detector was not operational at that time. Similarly, the TIRIS was not operational from 13 October 1998 to 23 December 1998 and so a visual detection rate of $\omega_t^{v} = 0.93$ was used during this time (Table 5.1).

Table 5.1. Number of tagged fish detected visually and electronically (TIRIS) at Macquarie Island (all grounds) and the detection rate by season. Only tags detected while the electronic detector was functional are included.

Season	Total tags detected	Detected both TIRIS & visually (n_{ve})	Detected TIRIS only	Detected visually only	Estimated no. recaptured (r_s)	Visual Detection rate (ω_i^v)	Detection rate (ω_i^{ve})
1996/97	80	36	11	33	90.08	0.77	0.89
1997/98	297	144	15	138	311.45	0.91	0.95
1998/99	40	13	1	26	42	0.93	0.95
1999/00	9	6	0	3	9	1.00	1.00

Catch numbers were estimated from the mean weight per fish from observer measurements and the daily total landed weight. A summary by season of the catch numbers, releases and recaptures in Aurora Trough and the Northern Valleys since the 1995/96 season are shown in Tables 5.2 and 5.3.

Table 5.2. Seasonal catch (numbers), release and recapture figures for the Aurora Trough region. Releases include re-released fish.

Season	Catch	Released	Recaptures							
			95/96	96/97	97/98	98/99	99/00	Win 00	00/01	Total
1995/96	180399	490	42	58	28	4	0	0	1	133
1996/97	79138	485		37	44	7	0	0	2	90
1997/98	45599	637			118	18	3	0	4	143
1998/99	12188	584				8	3	1	5	17
1999/00	1975	566					0	1	2	3
Win 2000	634	1						0	0	0
2000/01	9351	263							3	3
Total	329284	3025	42	95	190	37	6	2	17	389

Table 5.3. Seasonal catch (numbers), release and recapture figures for the Northern Valleys region. Releases include re-released fish.

Season	Catch	Released	Recaptures						
			96/97	97/98	98/99	99/00	Win 00	00/01	Total
1996/97	82909	540	3	53	5	0	1	0	62
1997/98	87953	538		76	9	0	0	0	85
1998/99	12902	312			9	0	0	0	9
1999/00	801	272				0	2	0	2
Win 2000	1623	2					0	0	0
2000/01	199	68						1	1
Total	186387	1732	3	129	23	0	3	1	159

Only 2 tagged fish have been recaptured after transferring between Aurora Trough and the Northern Valleys. A tagged fish released in Northern Valleys on 22 October 1997 was recaptured 87 days later on 17 January 1998 in Aurora Trough, and a fish released on 12

November 1997 in Aurora Trough was recaptured in the Northern Valleys on 22 January 1999 after 436 days-at-liberty.

Under a precautionary approach to the management of the population, the Aurora Trough ground was closed to commercial fishing for season 1998/99. However, a research quota of 40t was allocated to maintain the tagging program and population monitoring. With the reduction in quota, catches and recaptures were greatly reduced. However, a substantial number of tagged fish were released. The low catches and a high proportion of the releases occurring late in the season may explain the low number of recaptures from within the 1998/99 and 1999/2000 seasons.

Catches and releases began in the Northern Valleys region during January 1997, with 45 tagged fish being released on 1 January. During the first tagging season only 3 tagged fish were recaptured from a total of 540 releases (Table 5.3). These were recaptured on 18, 22 and 23 January 1997 after 16, 1 and 2 days free respectively. The low number of recaptures may be due to small catches being experienced after most of the tagged fish had been released, and the apparently large abundance of fish.

During the 1997/98 season there were another 538 releases in the Northern Valleys. Recaptures totalled 129, of which 53 were from 1996/97 tagged fish and 76 from within the 1997/98 season. The 1998/99 season saw a reduction in catch and a corresponding reduction in the number of recaptures. From a total of 312 releases, 9 were recaptured within the season. This trend continued during the summer 1999/2000 and June 2000 cruises, with poor catches and only 3 tagged fish recaptured. However, a substantial number of fish were tagged during the summer cruise (272), but only 2 fish were released during June 2000 (these were re-releases). Catch and recaptures in 2000/01 were very poor, with only 7 hauls made in this region in total (see Section 6). The recapture rate of 1 fish from 199 fish caught is much greater than in previous seasons, and clearly will result in a small estimate of available abundance (and large bounds on confidence limits). This highlights the large influence of recaptures with small sample sizes. There were an additional 10 hauls made in regions external to Aurora Trough and the Northern Valleys region during 2000/01.

Although there is no direct information on natural mortality of Macquarie Island toothfish, the known longevity of the species would indicate that natural mortality was less than $M = 0.2$ per year (Constable et al., 2001). In this report, natural mortality is assumed to be similar to that estimated for Heard Island Patagonian toothfish, namely $M = 0.1$ per year. A natural mortality value comparable to that for South Georgia toothfish, $M = 0.16$, was also considered to evaluate model sensitivity (SC-CAMLR, 1998; Constable et al., 2001). The growth curve parameters have been estimated by Constable et al. (2001) from model fits to age-length data for Macquarie Island Patagonian toothfish, and are: $L_\infty = 185.5$cm, $k = 0.042$, and $t_0 = -0.781$.

Constable et al. (2001) estimate the vulnerability at length of Macquarie Island fish with a linearly increasing and decreasing function illustrated in Figure 5.1 (parameters given in Table 5.4). The vulnerability function combines the effects of gear selectivity and fish availability. To account for observed changes in within season availability, Constable et al. (2001) estimate two vulnerability functions. The magnitude of the daily catch rate was assumed to be a reasonable indication of the availability of large or small fish. As such, the length composition of catch for which catch rates were less than 10 t/km^2 was used to estimate the vulnerability when only resident smaller fish were available. All length composition data were used to estimate a broader vulnerability curve. Within the assessment model, the observed daily catch rates being above or below 10 t/km^2 determines when the model applies each estimated vulnerability curve.

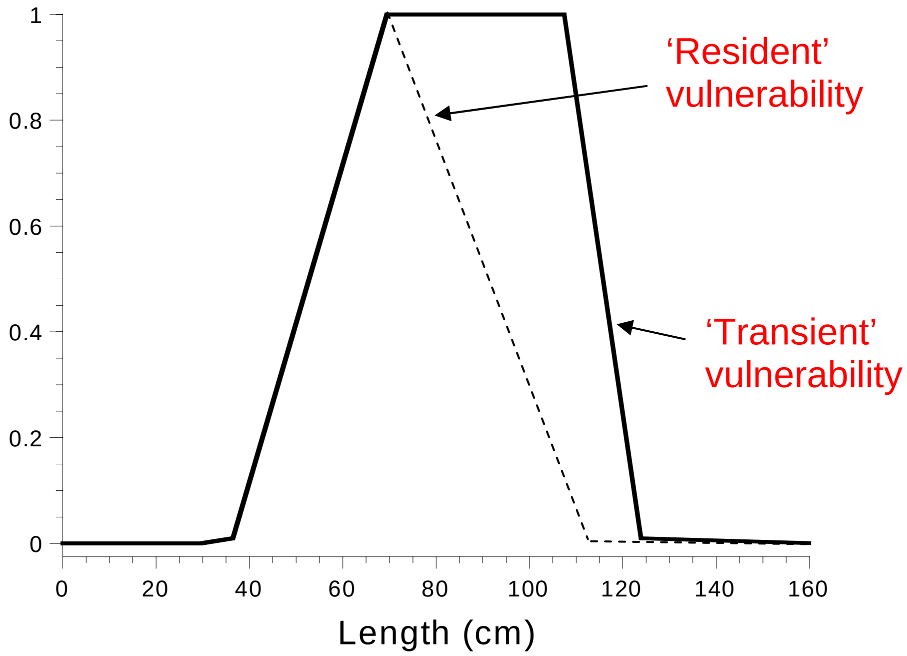


Figure 5.1. The vulnerability functions applied for Macquarie Island Patagonian toothfish. The dashed line represents the base vulnerability. The broader vulnerability curve incorporates the smaller fish of the base vulnerability and additional larger fish that become available if conditions are favourable.

Table 5.4. Vulnerability parameters as a function of length (cm) for Macquarie Island Patagonian toothfish estimated by Constable et al. (2001). Number pairs are the vertices of a linearly increasing and decreasing vulnerability curve as a function of length. The base-vulnerability curve has been estimated using only length-age data when catch rates are less than 10 t/km^2. The broad vulnerability has been estimated using all age-composition data.

	Length (cm)	Vulnerability
Base vulnerability	29.5	0.0
	36.5	0.01
	69.5	1.0
	113.4	0.006
	160	0.0
Broad vulnerability	29.5	0.0
	36.5	0.01
	69.5	1.0
	107.5	1.0
	123.9	0.01
	160	0.0

Aurora Trough

Figure 5.2 shows for the first season the proportion of recaptured tagged fish that remain in the analysis for the Aurora Trough region when the time to full mixing, δ, is increased. If there are no tagged fish remaining in the analysis (δ>78) then the model cannot estimate both pre-tagging available abundance and the 1996 net recruitment. The results presented below consider only δ = 0 (tagged fish mix immediately) and δ = 10. The choice of δ = 10 balances the need for adequate data for estimation (smaller δ values leave more tags for estimation; Figure 5.2) and allows time for tagged fish to mix (the larger δ, the more likely tagged fish will have mixed; see Tuck et al. (2000)).

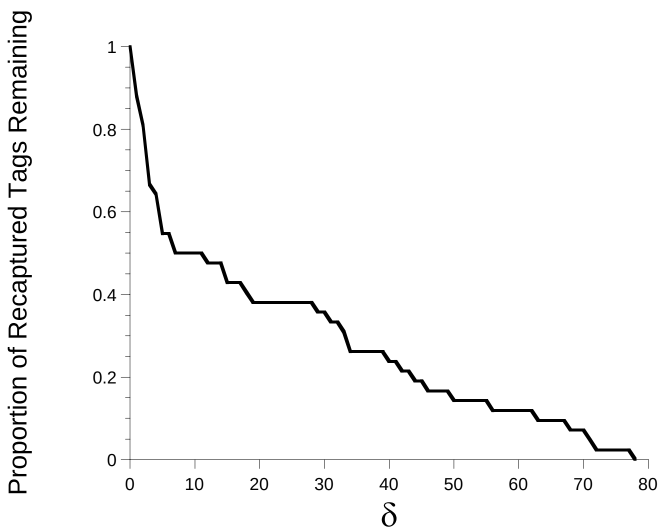


Figure 5.2. The proportion of tagged fish recaptured in Aurora Trough in the first season that remain in the analysis to estimate pre-tagging available abundance as a function of mixing level, δ. Of all the tagged fish recaptured during season 1995/96, the largest number of days-at-liberty was 78.

Point estimates and 95% confidence limits

For Models LIS and LDS, Tables 5.5 and 5.6 show the point estimates and 95% confidence limits for pre-tagging available abundance and net recruitments (millions of fish) for δ = 0 and 10, with M = 0.1. The increasing uncertainty in parameter estimates in recent seasons is clearly observed. The 95% confidence limits range over 1.5 million fish for the year 2000 net recruitment, whereas the range was only 0.2 million fish for the 1997 net recruitment. This increased uncertainty is due to the small number of catch records between 1999 and 2001.

Both models predict positive recruitment to the available abundance from 1998 onwards. Negative values of net recruitment suggest a net exodus of available fish between seasons. This exodus may have been due to emigration exceeding immigration (and local recruitment to the available population).

Table 5.5. Point estimates and 95% confidence limits for model LIS for Aurora Trough showing pre-tagging available abundance N_0 and annual net recruitments (millions of fish) for δ = 0 and 10, with M=0.1. The net recruitment in year 1996 is given by R_{96}.

LIS						
δ	0			10		
Parameter	Lower 95	Estimate	Upper 95	Lower 95	Estimate	Upper 95
N_0	0.58	0.72	0.94	0.68	0.94	1.41
R_{96}	-0.19	0.03	0.20	-0.59	-0.16	0.10
R_{97}	-0.21	-0.11	-0.03	-0.21	-0.10	0.00
R_{98}	0.10	0.22	0.39	0.12	0.26	0.49
R_{99}	-0.09	0.23	0.95	-0.20	0.14	0.83
R_{00}	-0.26	0.47	1.21	-0.10	0.65	1.58

Table 5.6. Point estimates and 95% confidence limits for model LDS for Aurora Trough showing pre-tagging available abundance N_0 and annual net recruitments (millions of fish) for δ = 0 and 10, with M=0.1.

LDS						
δ	0			10		
Parameter	Lower 95	Estimate	Upper 95	Lower 95	Estimate	Upper 95
N_0	0.47	0.58	0.74	0.54	0.73	1.07
R_{96}	-0.10	0.06	0.19	-0.38	-0.07	0.13
R_{97}	-0.16	-0.08	-0.02	-0.17	-0.08	-0.01
R_{98}	0.11	0.20	0.33	0.12	0.23	0.41
R_{99}	-0.08	0.16	0.71	-0.17	0.10	0.62
R_{00}	-0.09	0.49	1.14	0.05	0.65	1.48

Estimated annual available abundance

As the models keep an account of daily changes in available abundance for the duration of the tagging experiment, estimates of population trajectories can be displayed. For δ = 10, Figure 5.3 shows point estimates of available (a) numbers and (b) biomass for each model at 1 July 1995, 1 July 2001 and 1 January of each year in between. The estimate of pre-tagging abundance varies between 0.7 to 0.8 million fish for the selectivity models and is 0.94 million fish for model LIS. Estimated available abundance decreased to below 0.3 million fish during the 1997/98 season. However, since then available abundance has increased. The estimate of abundance on the final modelled day (1 July 2001) is between 0.8 and 1.0 million fish. The increase in available abundance corresponds well to the period where targeted commercial fishing ceased (a research quota of 40t was permitted). Figure 5.4 shows the 95% confidence limits about point estimates of available biomass for model LDS with M = 0.1 and δ = 10.

Available biomass was calculated by multiplying the daily available abundance by the average annual weight of fish caught in the Aurora Trough region (Table 5.7). Figures 5.3b and 5.4 show that the estimated available biomass declined from about 4,000-5,000t to approximately 1,000t during 1997/98. Since then the available biomass has increased to approximately 3,000t. Note that while the 1999 recruitment shows an increase in available abundance, the biomass shows a corresponding decrease due to the low mean weight per fish in season 1999/00.

Table 5.7. The mean weight (kg) by season of fish caught in the Aurora Trough region by daily catch estimation method.

Aurora Trough	1995/96	1996/97	1997/98	1998/99	1999/2000	Win 2000	2000/01
Mean Weight (kg)	5.16	3.53	4.11	4.32	2.9	3.3	2.75

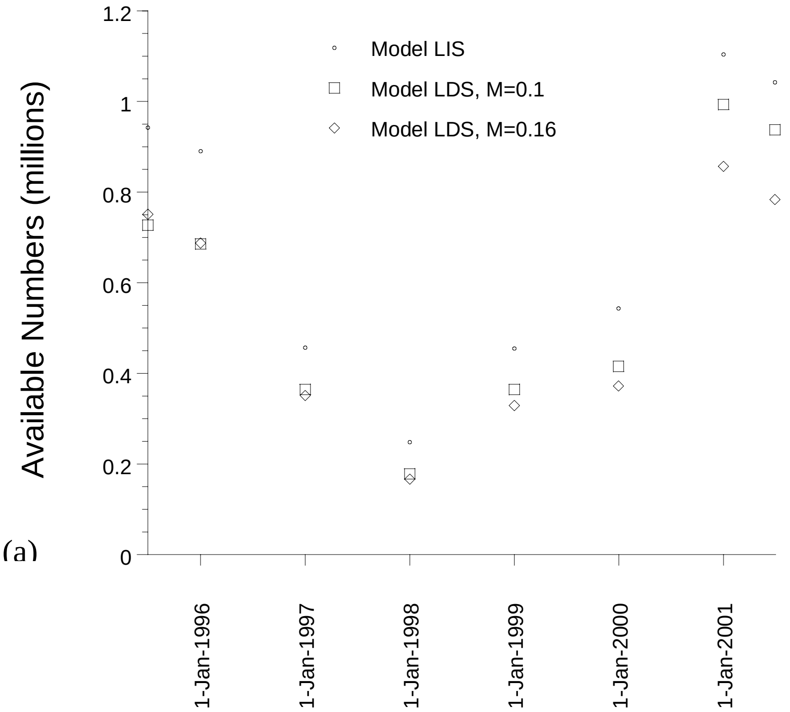
Abundance relative to pre-tagging levels

The percent of available abundance (numbers and biomass) at 30 June 2001 relative to 1 July 1995 are shown in Table 5.8 for the various models and for $\delta = 0$ and 10. Depending on the level of mixing and the particular assessment model, the estimates of final available abundance relative to the pre-tagging level vary between 104% and 139% in numbers and between 56% and 75% in available biomass. The available biomass does not show the same degree of increase due to the smaller mean weight per fish relative to the first season (Table 5.7).

As mentioned in the introduction, these figures relate to the abundance that is available to the fishery (i.e. the available abundance) and not to the total abundance of the population. It is important to note the large confidence limits associated with the 1999 and 2000 recruitment values, as this level of uncertainty reflects directly onto these estimates.

Table 5.8. The percent of available abundance (numbers and biomass) at 30 June 2001 relative to 1 July 1995 in Aurora Trough according to mixing level, δ, the assessment model and assumed natural mortality, M.

δ	0		10	
Model	Number	Biomass	Number	Biomass
LIS	126%	67%	110%	59%
LDS, M=0.1	139%	75%	127%	69%
LDS, M=0.16	115%	61%	104%	56%



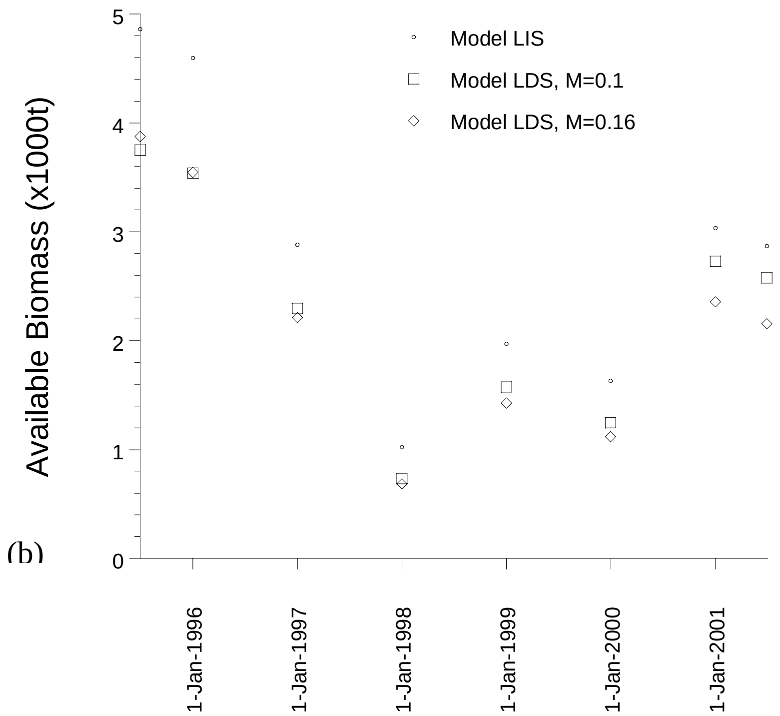


Figure 5.3. The predicted available (a) number and (b) biomass for Aurora Trough with $\delta = 10$ for assessment models LIS (the selectivity is equal to one for all lengths), and LDS (the selectivity changes according to the magnitude of the catch rate; Figure 5.1). Points shown are for 1 July 1995, 1 July 2001 and 1 January of each year between. These models assume $M = 0.1$ unless stated otherwise.

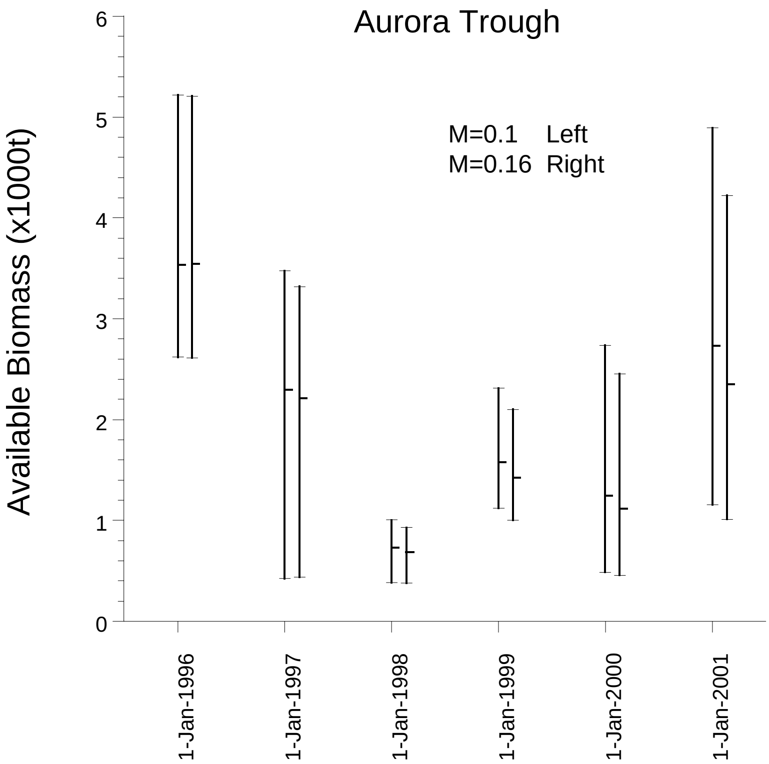


Figure 5.4. The 95% confidence limits about the point estimate of available numbers of toothfish at Aurora Trough at 1 January in each year since 1996 for model LDS with M =0.1 and 0.16, and δ = 10.

Northern Valleys

Point estimates and 95% confidence limits

Tables 5.9 and 5.10 show for models LIS and LDS the estimates of pre-tagging available abundance and net recruitment for the Northern Valleys region assuming M=0.1. As there were only three tags recaptured in the first season of tagging, the 95% confidence intervals are very broad. Estimates of pre-tagging available abundance are also large due to the small number of recaptures relative to the catch. The largest value of δ that can be considered is 16 as that is the longest duration of a tag being at-liberty before recapture in the first season of tagging. Estimates of 1997 recruitment show large negative values, indicating a large decline in estimated available abundance. The estimated level of recruitment in 1998 is approximately zero for all values of delta, while for 1999 it is approximately 0.5 million fish. The 95% confidence limits for the 1999 and 2000 net recruitments are very large. This is due to only three recaptures during the June 2000 cruise, none during the summer 1999/2000 season and only one in season 2000/01. Catch over this period was also extremely low (Table 5.3).

Table 5.9. Point estimates and 95% confidence limits for model LIS for the Northern Valleys showing pre-tagging available abundance N_0 and annual net recruitments (millions of fish) for δ = 0 and 10, with M=0.1. The net recruitment in year 1997 is given by R_{97}.

LIS						
δ	0			10		
Parameter	Lower 95	Estimate	Upper 95	Lower 95	Estimate	Upper 95
N_0	1.62	4.10	16.29	1.72	7.31	93.81
R_{97}	-3.51	-2.96	-0.71	-6.41	-5.81	-0.75
R_{98}	-0.17	0.05	0.37	-0.21	0.02	0.37
R_{99}	-0.19	0.52	3.60	-0.23	0.48	3.50
R_{00}	-3.51	-0.70	2.99	-2.67	-0.69	2.93

Table 5.10. Point estimates and 95% confidence limits for model LDS for the Northern Valleys showing pre-tagging available abundance N_0 and annual net recruitments (millions of fish) for δ = 0 and 10, with M=0.1.

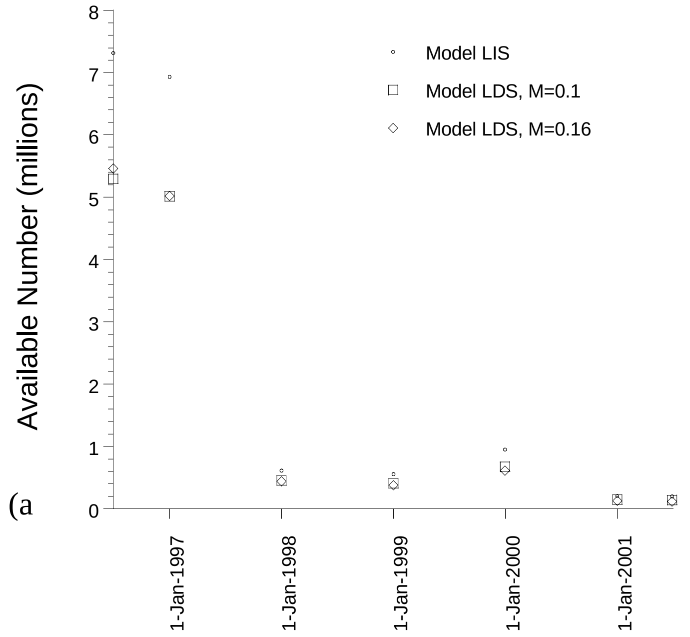
LDS						
δ	0			10		
Parameter	Lower 95	Estimate	Upper 95	Lower 95	Estimate	Upper 95
N_0	1.14	2.85	11.27	1.26	5.30	18.80
R_{97}	-2.37	-1.97	-0.42	-4.60	-4.15	-0.50
R_{98}	-0.13	0.03	0.26	-0.16	0.01	0.27
R_{99}	-0.14	0.36	2.56	-0.17	0.33	2.49
R_{00}	-2.49	-0.50	2.11	-2.44	-0.49	2.10

Estimated annual available abundance

Figure 5.5 shows the point estimates of available (a) numbers and (b) biomass for the two assessment models at 1 July 1996, 1 July 2001 and 1 January of each year in between. A large decline in available abundance (both numbers and biomass) is clearly evident from the first to second seasons. The biomass decline is exacerbated by a decrease in the mean weight of fish for the 1997/98 season (Table 5.11). It should be noted that this substantial decline could not have been due to fishing alone, as less than 90,000 fish were caught in season 1996/97 (Table 5.3) while the decline may have been up to 6 million fish (R_{97} of Tables 5.9 and 5.10). The models predict a final abundance of less than 0.25 million fish. The decline in available abundance between the seasons 1999/00 and 2000/01 is due to the recapture of a single tagged fish from very little catch; leading to a recapture rate that is much greater than previous seasons. Clearly the capture of the single tagged fish is having a strong influence on the results (see the confidence limits in Tables 5.9 and 5.10; Figure 5.6). Caution should be taken when interpreting the results from the most recent season.

Table 5.11. The mean weight (kg) by season of fish caught in the Northern Valleys region.

Northern Valleys	1996/97	1997/98	1998/99	1999/2000	Win 2000	2000/01
Mean Weight (kg)	6.03	4.35	3.14	1.58	3.22	1.55



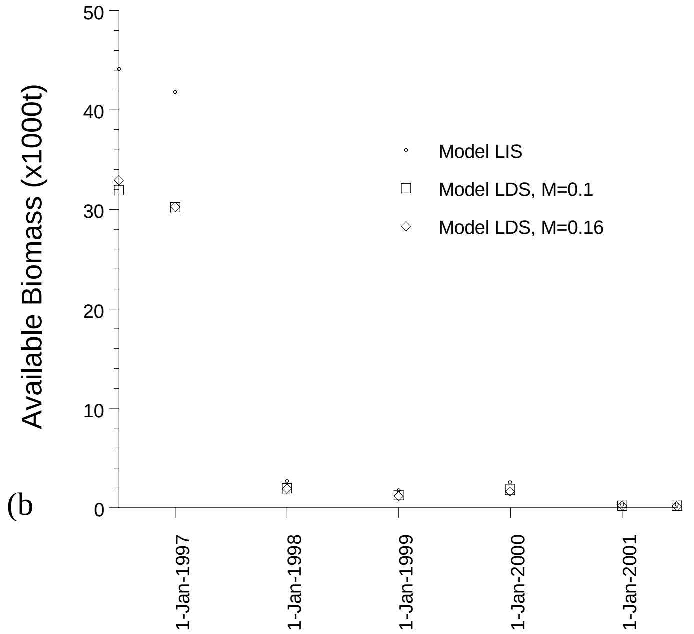


Figure 5.5. The predicted available (a) number and (b) biomass for the Northern Valleys with $\delta = 10$ for assessment models LIS (the selectivity is equal to one for all lengths and LDS (the selectivity changes according to the magnitude of the catch rate; Figure 1). Points shown are for 1 July 1996, 1 July 2001 and 1 January of each year between. These models assume $M = 0.1$ unless stated otherwise.

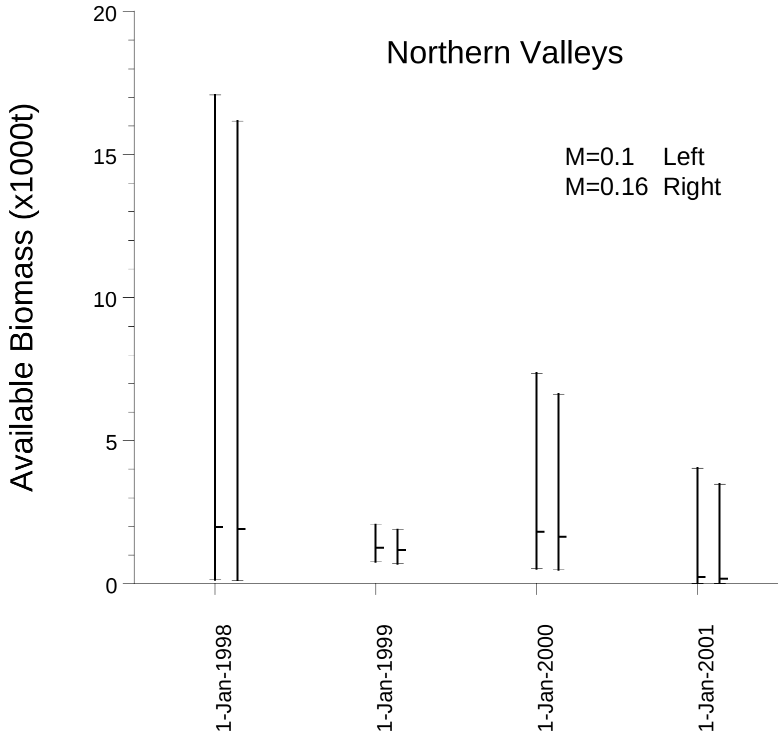


Figure 5.6. The 95% confidence limits about the point estimate of available biomass of toothfish for the Northern Valleys at 1 January in each year since 1998 for model LDS with M =0.1 and 0.16, and δ = 10.

Abundance relative to pre-tagging levels

The percent of available abundance at 30 June 2001 relative to 1 July 1996 has not been calculated for the Northern Valleys as the level of certainty in the final estimate of available abundance is so low as to make any statements about the degree of decline potentially misleading. This is compounded by the possibility that this region experienced an influx of non-resident fish during the first season. Comparing first season available abundance to current levels is not appropriate if this scenario is true without making some attempt to remove the 'transient' fish and model the abundance of the 'resident' stock alone (see Appendix C).

5.3.2 Discussion

Assessment models

The tag-recapture assessment model for Macquarie Island Patagonian toothfish is a novel modelling approach to the estimation of abundance of an exploited fish population where tag-recapture data provide the basic monitoring information. The model takes account of the exact time of catches, releases and recaptures. It readily allows for delayed mixing of the tagged fish with the general population. The data requirements of the model are rather stringent, particularly for daily catch numbers, which may not be practical for many fisheries. However, the Macquarie Island Patagonian toothfish fishery does meet these requirements.

Decreasing availability with length has meant that the catch from the trawl fisheries for Patagonian toothfish is generally composed of small, young fish (Williams et al., 1998; Constable et al., 1999). It is likely therefore that older, larger, tagged individuals are moving out of the available population (to deeper water for example). To account for this, an assessment model was developed that applied the estimated vulnerability curves to the probability of recapture. This model was referred to as the length-dependent selectivity (LDS) model. Other models that attempt to account for decreasing availability were developed, with details in Tuck et al. (1999, 2000) and Tuck and Williams (2001). When applied to the Macquarie Island toothfish fishery, the LDS model predicts a lower available abundance than does the LIS model. This is because the latter model assumes that all tagged fish, once released, remain in the available population until they are either caught or die of natural causes. Thus the number of released fish available to the gear is over-estimated and as a Petersen approach is used, where abundance is proportional to the number of marked fish in the population, an over-estimate of abundance results. The bias associated with estimating the number of marked fish in the available population (and hence the bias in the estimate of available abundance) is likely to increase over time as more tagged fish move out of the available population. The LDS model, explicitly defines the vulnerability function for the population and assume that the likelihood of recapture fluctuates with vulnerability. As a tagged fish ages and grows its vulnerability and hence probability of recapture changes accordingly. The model assumes that the vulnerability is narrower when catch rates are small. This is an attempt to model changing within-season availability. Occasionally, conditions are such that larger fish become available to the fishery. This leads to an increase in catch rates and increase in availability of big fish; including tagged fish.

The assessment models estimated that pre-tagging available biomass was between 3 and 5 thousand tonnes in Aurora Trough and between 30 and 45 thousand tonnes in the Northern Valleys region. Confidence limits on estimates of pre-tagging abundance for the Northern Valleys are very broad due to the recapture of only three tagged fish in the first season. This region shows a dramatic decline in available abundance between the first (1996/97) and second (1997/98) seasons. This reduction in available biomass is greater than can be explained by the fishery catches even in the absence of recruitment. For Aurora Trough, estimated available abundance declined over the first three seasons of tagging (to 1997/1998), but has since shown an increasing trend. This increase corresponds well to the beginning of catch restrictions for commercial fishing (a 40t research quota only was allowed) in the Aurora Trough region. For Aurora Trough, estimates of the fraction of current available abundance relative to pre-tagging levels are between 100% and 140% in numbers and 56% and 75% in terms of available biomass. Estimates of recent available abundance are most uncertain due to poor catches and the strong influence of a small number of recaptures (especially in the Northern Valleys, e.g. 1 tag recapture from 7 shots in 2000/01), and this uncertainty is directly reflected in any estimates of relative biomass.

Management implications

The dynamics seen in Aurora Trough would suggest the consistent presence of a stock, though with varying recruitment. However, in the Northern Valleys there has not been a significant catch since 1997. There are two different hypotheses as to why this could occur:

1) the one stock hypothesis:
all fish are part of a single stock, including those that may be regarded as resident on the trawl grounds and those that may be regarded as transient to the trawl grounds. An increased proportion of the stock periodically becomes available to the trawl fishery; and

2) the two stocks hypothesis:
there is a resident local stock in the Northern Valleys and a transient stock that is also occasionally available to the fishery. The resident stock is considerably smaller than the transient stock.

In 2001, the available biomass in Aurora Trough was estimated to be at or below the target level needed to allow commercial fishing to resume (SAFAG13, 2001). However, there was a large amount of uncertainty associated with these estimates as a result of the low number of trawls and the fact that there were only a few tagged fish recaptured in the most recent seasons. Table 5.12 shows that the point estimate and 95% confidence intervals for the most recent catch year (as at 1 July 2001) were 2154 tonnes (913 – 3886t), compared to estimates in 1995 of 3877 tonnes (2864 – 5676t). This indicates that the available stock was 55% of the initial abundance. This is still below the target level of reduction (to 66%) for the available biomass that is expected to lead to a long-term reduction in the spawning biomass to 50%.

Hence in 2001, SAFAG recommended that:

- the Aurora Trough region remain closed to commercial fishing and that a research TAC of 40t be set for the 2002 administrative season (SAFAG13, 2001).

Table 5.12. Point estimates and 95% Confidence Intervals of available biomass for Aurora Trough on 1 July 1995 (initial) and 1 July 2001 (last estimate) for assessment model LDS, M = 0.16 and δ = 10.

Aurora Trough (available biomass t)			
Date	Lower 95% CI	Point Estimate	Upper 95% CI
1-July-1995	2864	3877	5676
1-July-2001	913	2154	3866

In 2000, the TAC for the stock in the Northern Valleys was set at 420 tonnes (SAFAG10, 2000). This level was higher than the sustainable yield for the resident stock, however, it was considered acceptable as the catch rates in the fishery would not be high enough to make it economically viable to take this amount from resident fish. The TAC was set at this level to allow enough fish to be taken to trigger a higher TAC, which would apply if the transient stock reappeared. After re-examination in 2001, SAFAG determined that it was possible to trigger the higher TAC within the sustainable yield for the resident stock (SAFAG13, 2001).

In 2001, the situation in the Northern Valleys was largely unchanged from that reported in the previous year (SAFAG13, 2001). There was no indication that the transient fish under either hypothesis had returned since the 1996/97 season (Tables 5.13 and 5.14). Table 5.14 shows that the most recent reliable estimate (1 July 2000) of the available biomass of the resident stock was 1517 tonnes (448 – 6112t). Uncertainty still exists regarding the available biomass of resident fish in 1996, the initial year. Table 5.14 illustrates that, under the 2 stocks hypothesis, it is currently not possible to assign separate biomass estimates to resident and transient fish in 1996. To overcome this problem, two stocks assessment models are being developed (see Appendix C). However, taking the 1997 level as the best estimate of an initial available biomass of the resident stock, and the 2000 level as the best estimate of current available biomass, this is a reduction to 62%. This was considered to be consistent with the target level of depletion at which an annual sustainable catch is available (SAFAG13, 2001).

Table 5.13. Point estimates and 95% Confidence Intervals of available biomass for the Northern Valleys on 1 July 1996 and 1997 (initial) and since 1 July 2000 for assessment model LDS, $M = 0.16$ and $\delta = 10$ for the single stock hypothesis.

Northern Valleys – 1 stock hypothesis (available biomass)			
Date	Lower 95% CI	Point Estimate	Upper 95% CI
1-July-1996	**7821**	32928	164559
1-July-1997	472	**2427**	17889
1-July-2000	448	1517	6112
1-July-2001	NA	182	3207

Table 5.14. Point estimates and 95% Confidence Intervals of available biomass for the Northern Valleys on 1 July 1996 and 1997 (initial) and since 1 July 2000 for assessment model LDS, $M = 0.16$ and $\delta = 10$ for the two stocks hypothesis.

Northern Valleys – 2 stocks hypothesis (available biomass)				
	Transient Fish		Resident Fish	
Date	Pt Est	L 95% CI	Pt Est	U 95% CI
1-July-1996	?	?	?	?
1-July-1997	0	472	**2427**	17889
1-July-2000	0	448	1517	6112
1-July-2001	0	NA	182	3207

The recommended TAC was a constant annual yield of 10% of the best estimate of the initial available biomass of the resident stock, being 242 tonnes (10% of 1997 estimate). The available biomass is that available to the trawl fishery, and consists of a relatively restricted number of year-classes of mainly immature fish. Some population parameters were altered to be consistent with most recent CCAMLR assessments including using a natural mortality rate of 0.16.

To allow the transient stock to be fished, if it becomes available, SAFAG also recommended that a TAC of 782 tonnes be triggered if catch rates reach a threshold of an average catch rate of

10 tonnes/km^2 over 3 consecutive fishing days. This was a conservative TAC for the combined transient and resident stocks set at 10% of the lower 95% confidence interval of the abundance estimated to be initially present (i.e. in 1996; see Table 5.14). If catch rates were to fall below 10 tonnes/km^2 over 3 consecutive days, the TAC would revert back to 242 tonnes, or if more than 242 tonnes had already been taken, the fishery would be closed.

As such, in 2001 SAFAG recommended that:

- the toothfish TAC for the Northern Valleys for the period 1 January 2002 to 31 December 2002 be set at 242 tonnes, and

- a TAC of 782 tonnes be triggered if catch rates reach a threshold of an average catch rate of 10 tonnes/km^2 over 3 consecutive fishing days (SAFAG13, 2001).

5.4 Heard Island and McDonald Islands[4]

5.4.1 Results

Tagging of Patagonian Toothfish began in the Heard Island trawl fishery in May 1998 after the fishery itself commenced in April 1997 (Williams et al., 2002). From data on numbers of fish caught and numbers of tagged fish released and recaptured, estimates of available abundance were made for Heard Island's Ground B using the single-population assessment model described in Section 5.2. The model was only applied to Ground B because of the relatively low number or sporadic nature of recaptures in the other grounds. The fishing season is assumed to occur from 1-December to 30-November. Table 1 shows the seasonal catch, release and recapture information from May 1998 to June 2001. Recall that the parameters estimated by the model are the pre-tagging available abundance N_0 and the annual net recruitment, R_y (defined as the net change in available abundance between seasons). The potential impact of illegal, unreported or unregulated (IUU) fishing on the results has not been directly taken account of in this model. However, the loss of fish to IUU fishing would be indistinguishable, as far as the model is concerned, from migration, and will therefore appear as another factor within the net recruitment term (which includes juvenile recruitment and migration effects).

Table 1. Seasonal catch (numbers), release and recapture figures for Ground B of the HIMI fishery used in the tag-recapture assessment. Releases include re-released fish.

			Recaptures				
Season	Catch	Releases	1997/98	1998/99	1999/00	2000/01	Total
1997/98	1108435	558	21	52	22	5	100
1998/99	1112974	707		56	67	15	138
1999/00	1133052	782			115	65	180
2000/01	365624	1097				81	81
Total	3720085	3144	21	108	204	166	499

Fish caught in the HIMI trawl fishery generally fall between lengths 450mm to 1000mm (Constable et al., 1999). Fish vulnerability at Heard Island decreases with length because, it is believed, adults inhabit deeper water than recruited juveniles, which are outside the range of the fishery. The dome-shaped vulnerability functions estimated by Constable et al. (1999), and applied in this assessment model, allow for decreasing availability of larger and older fish. The assessment model assumes that the likelihood of recapturing a tagged fish of a given size is proportional to the vulnerability to fishing of that size class (Model LDS). Consequently, every released fish is aged using its recorded release length and estimates of von Bertalanffy parameters (L_∞ = 2465mm, k = 0.029 year^{-1}, t_0 = -2.56; data from SC-CAMLR, 2001). The tagged fish then grow each day according to the growth curve and, using the vulnerability function, the daily expected number of surviving available tagged fish is calculated.

As some uncertainty exists regarding the magnitude of natural mortality, M, for Heard Island Patagonian toothfish, two values covering the range of potential estimates are used to illustrate the sensitivity of the model to natural mortality. The chosen values are M = 0.1 and M = 0.16 (SC-CAMLR 2001).

[4] Text and figures adapted from Williams et al. (2002).

As with the Macquarie Island fishery, tagged fish are released in a group and so may not immediately mix with untagged fish, potentially causing a bias in parameter estimates. To account for this, the model allows a period of time for tagged fish to fully mix with the untagged population, δ days. For tagged fish recaptured within δ days of release both the release and recapture event are removed from the analysis. Parameter estimates were not sensitive to the mixing parameter δ over the wide range of values explored. However, the larger the value of δ, the less tagged fish remain in the analysis for estimation of parameters (and hence greater uncertainty in the estimates). A value of $\delta = 10$ is used in this analysis.

While released fish have been tagged with both plastic and electronic tags, not all of these fish will be detected on recapture. Unfortunately, the electronic detector has not always been functional and visual spotting is not perfect, so some tagged fish escape detection. An estimate of the tag detection rate ω can be made when both visual detection and the electronic tag detector are functional. A detection rate ω_{vis} based only on visual spotting of tags can also be calculated using this method. Noting that ω does not apply outside of the period when both detection mechanisms are possible, we have used the visual detection rate for all time periods, $\omega_{vis} = 0.9395$. As this will be lower than the overall detection rate it can be seen as a conservative estimate. Sensitivity to the application of this rate was considered by applying the joint detection rate over all time periods, $\omega = 0.9732$. Minimal changes to results were found.

On recapture, the number of T-bar tags remaining on the fish was noted in most cases. As all fish were double tagged, the number of recaptures where only one tag remained gives an indication of the tag loss rate. Of 682 fish observed, 79 or a proportion of 0.1158 had lost one tag. The model of Xiao (1996) was used to estimate the probability of losing both tags after one year. This model takes into account the likelihood that fish recaptured after a short time at liberty will have had less opportunity to lose a tag than those at liberty for a long time. From the proportion of fish that have lost one tag, the model concludes that the probability of losing both tags after one year at liberty is approximately 0.01.

Figure 5.7 shows estimates of available abundance on 1 July (the reference day) of each year of the fishery in Ground B with the 95% confidence limits (using the likelihood profile method) surrounding the point estimates. The results are not particularly sensitive to the applied natural mortality. Point estimates of available numbers decrease from approximately 7.5 million fish in 1998 to 3.5 million in 2001. It should be noted that these estimates lump different age groups of fish together, and so a continuous trajectory of available abundance over each season should not be inferred. This is because vulnerability has moved toward larger fish over the duration of the fishery (SC-CAMLR, 2001).

Predicted abundances of total fish from the projections using the Generalised Yield Model (GYM) (Constable and de la Mare, 1996; Williams et al., 2002) show similar trends to the results of the mark-recapture analysis, although the decline over the period 1998 to 2001 is not as great.

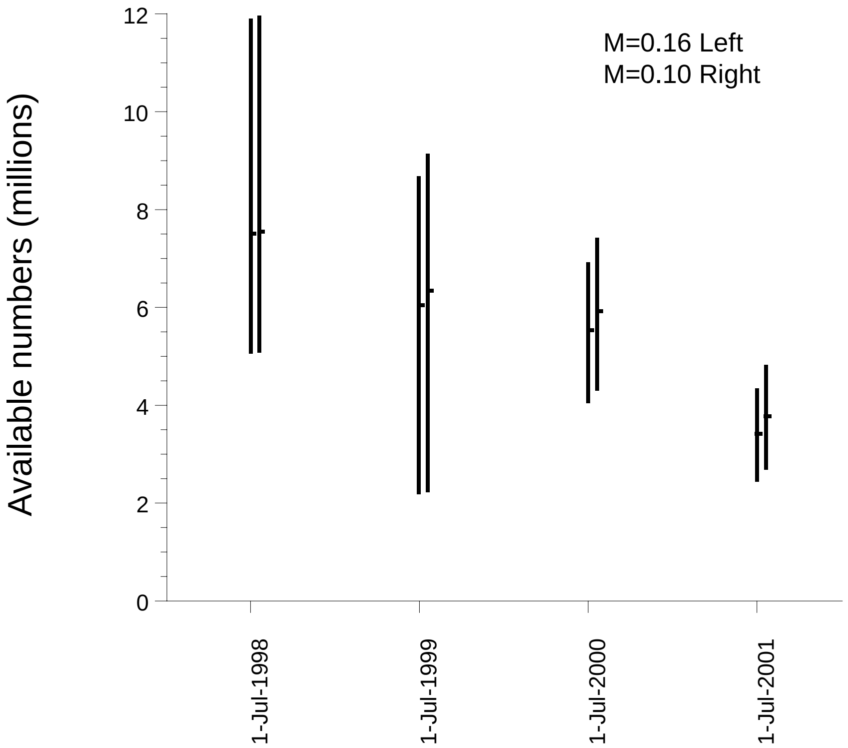


Figure 5.7. Estimated available numbers of fish with 95% confidence limits for Ground B in the Heard Island toothfish fishery with 1-July as the reference day. Shown are results with M=0.16 and M=0.10, and δ=10.

5.4.2 Discussion

The estimates of available numbers of fish in Ground B have declined steadily during the course of the experiment and are now at just under 50% of the value when the tagging program started. Recall that the tag-recapture method of stock assessment is only able to estimate available abundance. Available abundance is quite distinct from total abundance, as estimated by the Generalised Yield Model (GYM) (Williams et al., 2002). The projections using the GYM and based on the time-series of estimated annual recruitments of Age 4 fish since 1986 indicates that the overall population may have changed in a similar way as the local stock at Ground B. Ground B may have experienced a greater reduction in abundance than the overall stock because the fishery concentrates on the younger fish and it will be a few more years before the effects of the fishery are fully transmitted to the spawning stock. In addition, the greater catches from Ground B than other areas mean that the reduction in this area would be expected to be greater than for other areas and the whole stock.

The influence of IUU fishing on the use of the Petersen equation to estimate available biomass is difficult to determine. The main effect of IUU fishing on the model will be the over estimation of the number of tagged fish remaining in the water. This would lead to an over estimation of available biomass. The total estimated IUU catch for the 1998/99 to 2000/01 split years in the HIMI area is 2609 tonnes (SC-CAMLR, 2001) which is 26.6% of the 9795 tonnes of legal catch in the same period. Some of the IUU vessels have been sighted in or close to fishing grounds B and C, which suggests that some of their catch is from the same population as the legal catch. However, the proportion of catch taken in the same grounds as the legal fishery is not known. It can be expected that IUU catches of this magnitude will have an effect on the calculation of available biomass but there are insufficient data to estimate its magnitude.

The tagging experiment at Heard Island provides invaluable information about the dynamics of the fish inhabiting the area. The ability to estimate the available abundance using tagging data complements and further enhances the assessment methodologies (such as the Generalised Yield Model) already developed to manage the fishery (SC-CAMLR, 2001). Tagging programs are especially useful in providing available biomass estimates in areas where trawl surveys are difficult due to unsuitable bottom conditions (Macquarie Island) or where the commercial fishery relies on longlining, thus making fishery-independent surveys impractical. As the trends in population numbers from the mark-recapture analysis are similar to the trends predicted from the Generalised Yield Model, the potential exists to refine the assessment process. With further development and validation, the mark-recapture analysis may be able to be used to statistically weight the population projections used in the Generalised Yield Model in the same manner that CPUE is used in refining the assessments for toothfish at South Georgia (Kirkwood and Constable, 2001).

6. CATCH RATE MODELS OF RELATIVE ABUNDANCE

Geoff Tuck, Xi He and Tim Lamb

6.1 Introduction

This section presents an exploration and analysis of the catch and effort data obtained from the Macquarie Island fishery in order to provide a seasonal index of relative apparent abundance from a standardisation of catch rate data. The methods follow from the catch rate analyses of Tuck and Campbell (1999), Tuck (2000) and Tuck and He (2001). The data were provided from the catch and effort database maintained by the Australian Antarctic Division (Kingston, Australia).

6.1.1 Data summary

A summary of the catch and effort data used in this analysis is presented in order to gain further insights into the operational characteristics of the fishery and assist the development and interpretation of the standardised catch rates estimated later in this section. The data consists of individual records for each trawl and provide information on the following:

- location (Aurora Trough, Northern Valleys)
- season
- date (day, month, year)
- catch (tonnes)
- effort: swept area (km^2)
- fishing depth (metres)
- skipper (coded 1 to 3)

Data from Colgate Valley, Beer Garden and Grand Canyon were combined into the single location given the title Northern Valleys. The within-location sub-regions have been aggregated due to their relative proximity and tagging evidence that suggests a reasonable level of between region mixing. However, Aurora Trough and the Northern Valleys have been separated for this analysis, as evidence from genetic studies and tagging (only two tagged fish having moved between the regions) suggests that each of these populations may be considered as a single unit for management purposes. Areas of exploration outside of these locations (i.e. New Grounds) have not been included in the analysis. Data from the first season (1994/95) was also not included in the analysis as the majority of shots were exploratory and the main grounds were yet to be found. The data for Aurora Trough extends from 1995/96 to 2000/01. The Northern Valleys were not discovered until the 1996/97 season. A summary is also provided for the winter 2000 cruise, however these data are not included in the standardisation. Note also that the tagging program began in Aurora Trough during 1995/96.

The data were examined for erroneous records such as those with no effort recorded. There were 12 observations with very small effort recorded (swept area < $0.01km^2$) and these were removed from the analysis. Extremely large catch rates were observed in both areas during their first season. The influence of these records is examined by filtering out the high catch rate records, as they may have undue influence on results and could potentially be the result of human error (e.g. accidental miss-reporting or data punching errors). The months of operation range between October and March in summer, with a single winter cruise in June 2000 (see Table 6.1 and 6.2). Records for March are only observed in season 1995/96 in Aurora Trough. Unbalanced data sets can lead to poor estimates of parameters and biases in indices of abundance. As such, the month of March and the winter cruise were removed from the catch rate standardisation. Likewise, as records for months October and November are only found in seasons 1997/98 and 1998/99, these months have been combined into a single level for the analysis.

Table 6.1. The total number of records by season and month in Aurora Trough with average catch rate (t/km^2) and standard deviation. Records for the months of October and November have been combined in column Oct/Nov.

Month Season	Oct/Nov	Dec	Jan	Feb	Mar	June	n E[CPUE] StD[CPUE]
95/96		25	58	82	116		281
		1.46	109.8	51.0	5.65		40.0
		0.72	202.0	49.3	14.23		103.7
96/97		47	53	43			143
		28.95	23.58	17.85			23.6
		24.77	28.70	19.17			25.1
97/98	54	3	38	6			101
	16.13	2.37	10.03	7.04			12.9
	22.6	2.97	11.72	7.29			18.4
98/99	9	9	32				50
	7.1	5.13	12.6				10.2
	10.9	7.8	19.9				17.1
99/00			13	10			23
			1.9	0.65			1.4
			4.8	0.9			3.6
Winter						5	5
2000						1.4	1.4
						1.0	1.0
00/01		2	11				13
		5.33	13.95				12.6
		2.13	9.05				8.9
Monthly n	63	86	205	141	116	5	616

Table 6.2. The total number of records by season and month in the Northern Valleys with average catch rate (t/km^2) and standard deviations. Records for the months of October and November have been combined in column Oct/Nov.

Month Season	Oct/Nov	Dec	Jan	Feb	June	n E[CPUE] StD[CPUE]
96/97			65 248.68 613.44	16 1.84 3.83		81 199.92 557.51
97/98	29 4.9 7.3	146 28.8 82.7	88 7.0 14.4	13 3.6 8.4		276 18.11 61.7
98/99	24 0.3 0.4	40 1.43 1.42	40 3.6 7.6			104 2.0 4.95
99/00			16 0.26 0.91	10 0.07 0.10		26 0.18 0.7
Winter 2000					19 2.15 3.33	19 2.15 3.33
00/01		3 0.29 0.24	4 0.21 0.14			7 0.24 0.18
Monthly n	53	189	213	39	19	513

Tables 6.3 and 6.4 show catch rates (tonnes per km^2) over all seasons for both Aurora Trough and the Northern Valleys. The distribution of catch rates is clearly very broad, with the majority of records showing catch rates below 20 t/km^2, but with a substantial number of records in the hundreds and even thousands of tonnes per km^2 during the initial seasons of the fishery.

Table 6.3. The catch rate (t/km^2) by season for the Aurora Trough region. The number in a cell represents the number of records with a catch rate between that row and the previous row's catch rate, e.g. there were 29 records with a catch rate between 1 and 5 t/km^2 in season 96/97.

CPUE (t/km^2)	94/95	95/96	96/97	97/98	98/99	99/00	Jun-00	00/01
0	23	6	1	2	8	8	1	
1	41	10	4	1	14	9		
5	158	112	29	32	8	5	4	3
10	40	25	22	27	6			4
15	12	9	14	14	1			1
20	5	12	15	9	4	1		2
25	2	9	9	6	2			2
50	1	31	29	6	5			1
100		35	16	2	2			
150		15	4	2				
200		8						
300		7						
400		1						
400+		1						
Hauls	282	281	143	101	50	23	5	13

Table 6.4. The catch rate (t/km^2) by season for the Northern Valleys. The number in a cell represents the number of records with a catch rate between that row and the previous row's catch rate, e.g. there were 3 records with a catch rate between 300 and 200 t/km^2 in season 97/98.

CPUE (t/km^2)	96/97	97/98	98/99	99/00	Jun-00	00/01
0	19	15	19	12	1	1
1	10	78	43	13	9	6
5	11	71	35	1	7	
10	3	27	4		1	
15	6	26	1		1	
20	1	8	1			
25	1	12				
50	6	20	1			
100	6	9				
150	3	4				
200						
300		3				
400	3	2				
500	3					
1000	5	1				
2000	2					
4000	2					
Hauls	81	276	104	26	19	7

The fishing depth field is defined as the greater of the recorded start and finish depths. While not giving an exact measure of the depth at which fish were caught, it should give a reasonable measure of the relative difference in depth between hauls. In Aurora Trough the main fishing depth lies between 700m and 1000m. For the GLM analysis the depth factor was stratified into 3 levels according to catch rate trends, namely D<700m, 700≤D<900, D≥900m. Figure 6.1 shows the catch rate as a function of depth in Aurora Trough. The Northern Valleys show two main depths at which fishing occurs, namely between 650m and 800m and another mode between 1100 and 1300m (Figure 6.2). For the GLM analysis, depth was stratified into depths, D ≤ 1000m, and D>1000m.

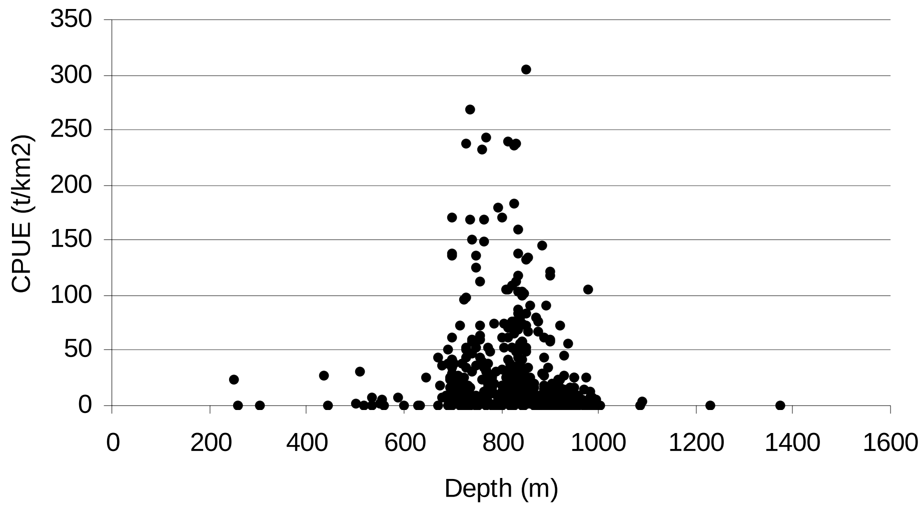


Figure 6.1. The catch rate (t/km^2) at Aurora Trough as a function of depth. One record with a catch rate of 1504 t/km^2 and depth 723m is not shown in order to enhance the resolution of the presented figure.

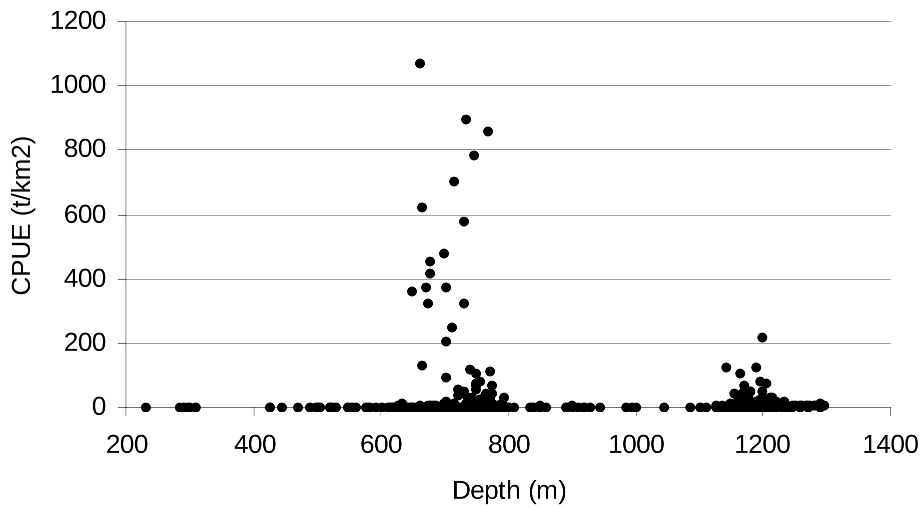


Figure 6.2. The catch rate (t/km^2) of toothfish from the Northern Valleys as a function of depth. Three records with a catch rate greater than 1200 are not shown in order to enhance the resolution of the presented figure. These catch rates (with depth) are 1900 (658), 2147(633), and 3861(639).

There have been 3 skippers during the 6 seasons since 1995/96. Unfortunately, not all have participated in the fishery in each season or month (Tables 6.5 and 6.6). The highly unbalanced nature of the data set in this instance reduces the power of the analyses to determine the relative influence of different skippers on fishing success. While skippers (and mates) may well be a significant contributing factor to observed and standardised catch rates, caution should be taken when interpreting results with skipper included as a model effect.

Table 6.5. The number of records for each skipper (A, B and C) in each season at Aurora Trough (AT) and the Northern Valleys (NV).

	AT			NV		
Season\Skipper	A	B	C	A	B	C
95/96	0	116	165	-	-	-
96/97	0	100	43	0	65	16
97/98	0	54	47	0	29	247
98/99	9	0	41	24	0	80
99/00	23	0	0	26	0	0
June 2000	0	0	5	0	0	19
00/01	0	0	13	0	0	7

Table 6.6. The number of records for each skipper (A, B and C) in each month at Aurora Trough (AT) and the Northern Valleys (NV).

	AT			NV		
Month\Skipper	A	B	C	A	B	C
Oct/Nov	9	54	0	24	29	0
Dec	0	47	39	0	0	189
Jan	13	53	139	16	65	132
Feb	10	0	131	10	0	29
March	0	116	0	0	0	0
June	0	0	5	0	0	19

6.2 Methods

Catch rate data are frequently used to obtain a relative measure of apparent abundance of a harvested stock. For this purpose, catch rates are assumed to be proportional to the density of the sampled population. While raw or nominal catch rates (total catch divided by total effort) can be used, undue weighting of high effort cells may bias the indices produced. In addition, changes in the fishery or the environment may strongly influence catch rates in any particular year. Hence, changes in catch rates may not be due to changes in stock abundance alone.

The standard method used to account for these biases and changes in the fishery is General Linear Modelling (McCullagh and Nelder, 1989). Factors that may influence catch rates can be either continuous (e.g. sea surface temperature) or categorical (e.g. skipper). The models in this report have season, month, depth and skipper as categorical effects. A month factor is included, as some months appear to produce larger catch rates than others (January, in particular). The models considered are:

Model 1: Season only

Model 2: Season, Month, Depth, Skipper

The general form of the log normal model is,

$$\ln(CPUE) = \ln\left(\frac{C_{s,m,d,k}}{1000 * E_{s,m,d,k}}\right) = \mu + \alpha_s S + \alpha_m M + \alpha_d D + \alpha_k K + \varepsilon \tag{6.1}$$

where $C_{s,m,d,k}$ is catch (kg) for season s, month m, depth d, and skipper k, a catch of 1.0kg was added to C if C≤1.0 so that the logarithm value can be calculated,

$E_{s,m,d,k}$ is effort (km^2, area swept) corresponding to the catch,

μ is the intercept,

α is the factor for each of the terms in the model: season (S), month (M), depth (D), and skipper (K), and

ε is an error term assumed to be independent, normal random variables with zero mean and constant variance, $N(0, \sigma^2)$.

Sensitivities to filtering the data for extreme catch rate values with potentially large and undue effects on model results are also explored. Two filtering schemes are considered: (i) all records by location are used, and (ii) only records where CPUE<200t/km^2.

The seasonal indices of abundance (I_s) relative to the final year of fishing are calculated using the following formula:

$$I_s = e^{(\alpha_s - \alpha_f)} \tag{6.2}$$

where α_f is the parameter estimate for the final season (2000/01).

6.3 Results for the Macquarie Island fishery

Figures 6.3 and 6.4 show the standardised catch rates for the Aurora Trough and Northern Valleys regions. Both regions show marked declines in the index of relative abundance in the first seasons of the fishery. In particular, the Northern Valleys show a decline of between 35 and 500:1 between season 1996/97 and 1999/00 depending on the model. However, the last season’s index shows an increase in both regions. It should be noted that since the beginning of the fishery, the number of records per season has decreased. The last season’s estimation is based on only 13 records in Aurora Trough and 6 in the Northern Valleys. The poor sample sizes leads to further uncertainties with regard to these indices and caution should be taken in any interpretation of the results.

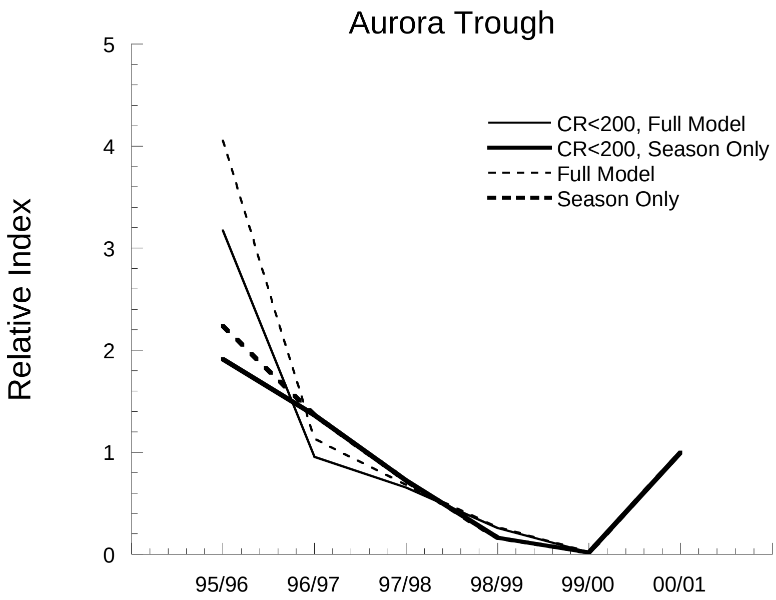


Figure 6.3. The relative indices of abundance from a standardisation of catch and effort data from the Aurora Trough region of Macquarie Island. Indices are shown where only catch rates less than 200 were used (CR<200) and where no catch rate filter is applied. The Full Model includes season, month, depth and skipper effects.

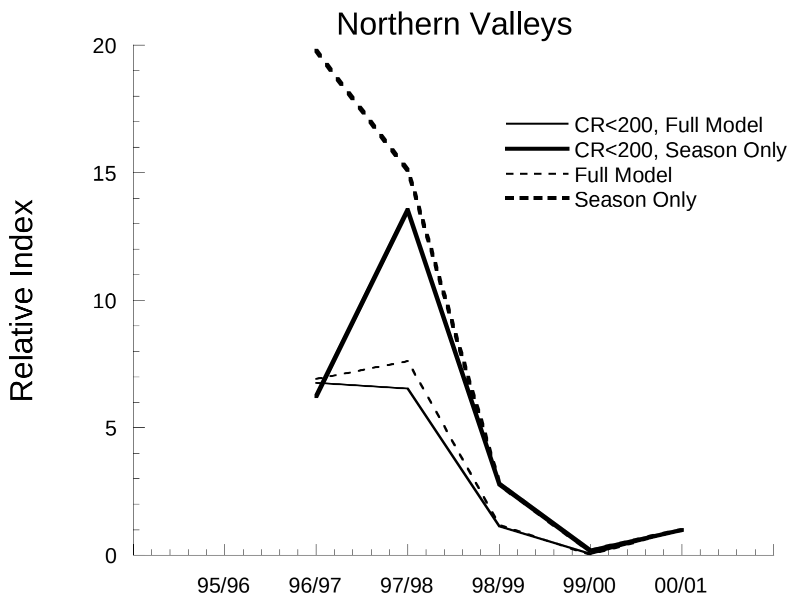


Figure 6.4. The relative indices of abundance from a standardisation of catch and effort data from the Northern Valleys region of Macquarie Island. Indices are shown where only catch rates less than 200 were used (CR<200) and where no catch rate filter is applied. The Full Model includes season, month, depth and skipper effects

For the Aurora Trough, the skipper effect was not as significant as the other model factors, being only marginally significant at 7.6%, as compared to other factors, which are all <0.01%. If the skipper effect is removed from the analysis, the indices show a similar rate of decline between the first and last season as the models with the skipper effect, however the season effect is highly significant. For the Northern Valleys region, the model without the skipper effect shows a large decline in the relative abundance index between the first and second seasons, whereas the model with the skipper effect shows a small increase. The skipper effect is highly significant and appears to be explaining much of the decline in observed nominal catch rates (especially between the first two seasons). The large changes in the nominal catch rates are being attributed to the effect of differences in the fishing success of the various skippers. As previously mentioned, as the data for skipper is very unbalanced and in some instances provides little contrast, these results and their interpretation should be treated with due caution.

6.4 Discussion

The analyses suggest that there have been substantial declines in available abundance of Patagonian toothfish in both of the major fishing grounds. The level of the decline is uncertain as the indices are sensitive to the degree of filtering, and the factors included in the models. This sensitivity is likely to be due to several factors, including (a) the short duration of the fishery (b) the unbalanced nature of the data, providing little contrast between modelled factors, (c) confounding between skipper and season/month effects, (d) the large variation in catch rates between seasons and months, (e) the influence of large catch rate values in the data, and (f) the small sample sizes in the most recent seasons.

7. MANAGEMENT STRATEGY EVALUATION[5] AND BELIEF MODELLING

Sakari Kuikka, Geoff Tuck, Keith Sainsbury, Tony Smith and Xi He

7.1 Management Strategy Evaluation

7.1.1 Introduction

Assessments of the population of toothfish at Macquarie Island have indicated that the population has undergone major changes in fishable abundance since the fishery began in the summer of 1994/95 (Sections 5 and 6). The spatial and temporal dynamics of the population remains largely uncertain. For example, it is unclear whether the population at Macquarie Island is a single well-mixed stock, or is composed of two or more local populations, or whether transient fish having a more cosmopolitan habit occasionally joining the resident fish. To a large extent, several other key components of the fishes' biology also remain uncertain. These include its reproductive biology, growth, natural mortality and the relationship between fishable abundance and total abundance (Constable et al., 2001). The challenge for managers is to establish methods for managing the harvested populations that recognise these uncertainties, and which satisfy, to a reasonable degree, various, often conflicting, management objectives

Management objectives for the Macquarie Island Patagonian toothfish fishery were discussed at a workshop attended by managers, industry and scientific representatives in April 2000 (Tuck, 2000a). Management goals included both conservation and utilisation objectives. Likewise, management strategies for the fishery have been discussed at various meetings of the Sub-Antarctic Fisheries Assessment Group. A management strategy is a set of pre-agreed rules for selecting management actions, designed to achieve specified management objectives. Apart from the objectives, the components of a management strategy are the sampling program, the assessment and a decision rule that translates the data from the sampling program and information from the assessment into a TAC.

The method used to evaluate these strategies is called Management Strategy Evaluation (MSE). MSE does not attempt to find an 'optimal' strategy, but rather explicitly outline the trade-offs inherent in managing stocks with potentially competing objectives. The aim of MSE is to consider alternative management strategies and examine their performance against a range of management objectives under various assumptions that encompass the system's uncertainties.

The basic method for evaluating performance is Monte Carlo simulation; an insightful mechanism to explore complex systems. This method involves the simulation of the fishery from its beginning to a pre-determined future year (2050 in this instance). An 'operating model' is used to simulate the 'true' dynamics of the toothfish population and the fishery, to generate future catch, tag releases and tag returns. An assessment model is applied to both historical and future (simulated) data to give an estimate of the status of the population (which is known through the operating model). Feedback management strategies can then use the annual

[5] This work was initiated under FRDC project 97/122. As many of the results reported in Chapter 14 of He and Furlani (2001) were completed under project 2000/109, only a summary of the methods and results are presented here.

estimated current status to set catch quotas. Once a scenario[6] and a management strategy have been chosen for evaluation, the biology and fishery dynamics are simulated (50 times). A range of performance statistics is produced for each simulation and management strategy/scenario combination. The performance statistics are combined over all simulations and tabulated to provide a summary of the performance of the particular management strategy for a given scenario. These summaries provide a means of comparing the performance of each management strategy across many scenarios.

The MSE software can be used to illustrate the potential effects of management strategies given particular assumptions about resource dynamics. The results, while presented quantitatively, should be interpreted qualitatively. The model is used to compare the decision alternatives (management strategies), not to predict exact future catch or population biomass. A special, and a dominant, feature of the model is that it includes vessel reaction to changes in catch rates. This model component was included to assess the effects of behavioural elements of the fishery.

This section gives an overview of the MSE analyses presented in Tuck et al. (2001b) and is followed by an application of Bayesian Belief Networks to the Macquarie Island fishery, which stems from the MSE work.

The general methodology used to evaluate management strategies is similar to that applied by Smith et al. (1996) and Polacheck et al. (1999). There are five main components:

1. An operating model that simulates the population and fishery dynamics in both historical and future years
2. A sampling model that generates the data available for assessing the resource from the “true” state of the resource as simulated in the operating model
3. An assessment model that uses the data from the sampling model to provide estimates of resource status
4. A harvest strategy component that determines management actions based on the results of the assessment model and/or specified decision rules. Note that some harvest strategies do not include yearly assessments.
5. A component for the calculation of an appropriate set of performance statistics

The first four components are sequentially iterated to simulate a time series of future population sizes, management actions, and catches. Each simulation has different random values that influence (a) daily variation in catchability, (b) the daily level of effort, (c) the daily number of released and recaptured fish, (d) the annual transient biomass, (e) annual observation error in mean weights of fish, and (e) annual fluctuations in recruitment. However, for efficacy of comparison of performance measures under differing management strategies and/or differing scenarios, the realised random variables for effort, catchability, recruitment, observation error and the transient biomass remain constant across management strategy/scenario combinations. For example, for a particular simulation and particular year within that simulation, the random variable determining recruitment will remain the same across all management strategy/scenario combinations. The results can then be used to evaluate the performance of a particular management strategy for a specific set of assumptions about the dynamics of the resource without doing large numbers of replicate trials to get unbiased results for comparison.

[6] A scenario is a specific set of assumptions about the dynamics of the resource. The scenarios can represent the model or parameter uncertainties of the system.

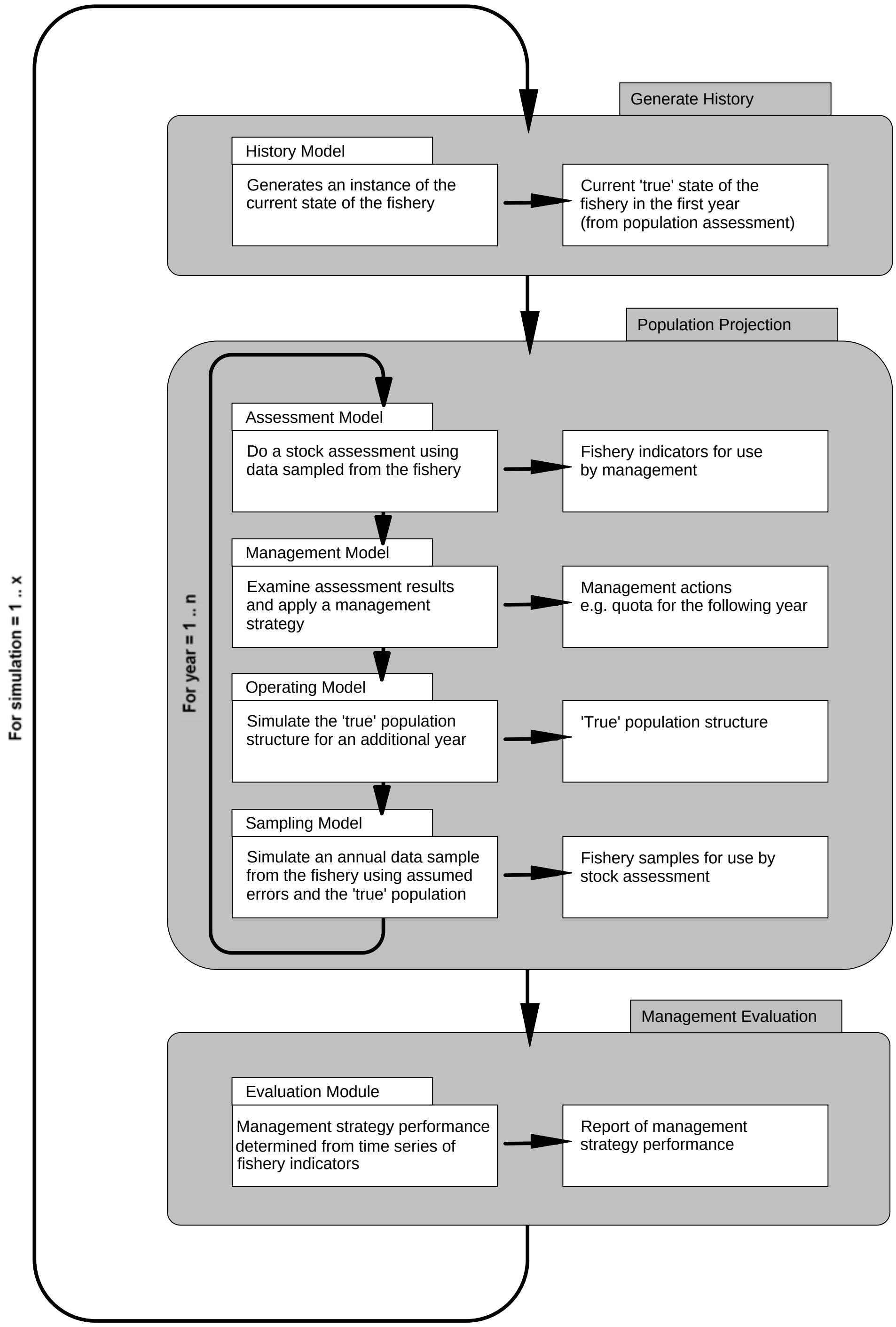


Figure 7.1 A diagrammatic representation of the management strategy evaluation framework for Macquarie Island toothfish (from Polacheck et al. (1999)).

Figure 7.1 provides a diagrammatic representation of the management strategy evaluation framework. Each of the MSE components for the toothfish management strategy evaluation is broadly described below.

7.1.2 Management objectives

The first requirement of MSE is a clear definition of the management objectives so that performance measures can be developed and used to compare and evaluate alternative harvest strategies. The agreed objectives were (Tuck, 2000a):

a) Maximise the discounted expected net returns over all simulated years.

b) Ensure catches and catch rates are above some minimum viable level each year.

c) Minimise the probability that the stock will fall below various levels relative to the unfished biomass.

d) Reduce levels of uncertainty in assessments.

e) Provide cost-effective levels of monitoring to achieve management objectives.

Objective (c) was further refined to give specific limits. The lower limit of long-term spawning stock biomass was agreed to be 50% of the unfished biomass, which corresponds to a reduction to 66% of the trawl available year-classes.

7.1.3 The biological component of the operating model

The operating model simulates the population and the fishery dynamics on a daily basis. The biological model of the toothfish population is a sex and age-structured dynamical population model based on standard catch and population dynamic equations. However, alternative models were developed in order to account for specific ecological scenarios regarding the fishery at Macquarie Island. Where possible best estimates of population parameters were applied and where key uncertainties existed plausible alternatives were used for sensitivity analyses.

Large inter-annual variability in available abundance of toothfish occurs at Macquarie Island. Three population models were developed to represent the possibility of the occasional influx of fish onto the fishing ground. These are:

i. A single stock model, where occasionally large fish occur in the catch due to a change in the size specific availability of a resident stock.

ii. A single stock model, as in (i) but with multi-year periods of zero recruitment when no young fish are recruited to the population

iii. A two stock model, where a second transient stock of fish occasionally moves into the region to join a smaller resident stock.

The first model assumes that there is only one well-mixed stock of fish being harvested, and that for some reason (e.g. environmental variability) larger fish from this stock occasionally become available to the fishery. All mature fish in this single stock contribute to reproduction, whether or not they are available to the fishery. The second model attempts to account for the observed poor recruitment in this fishery in some years, and allows for periods (years) where there has been complete reproductive failure, i.e. no zero year old fish are produced (or survive) to join the population. The third model assumes that there are two stocks of fish being harvested, one that is resident on the fishing ground and one that is non-resident or transient. Fish from the transient stock occasionally move into the region occupied by resident fish and can be caught and tagged there. These fish are reproductively isolated from the resident fish. These resource models vary considerably in their assumptions and one of the aims of management strategy evaluation is to provide management strategies that are robust (in terms of performance) across all resource models and other key uncertainties.

7.1.4 The fishery component of the operating model

The fishery simulator models the daily processes of catch, tagging, recapture and the behavioural dynamics of the fishing operation. For each day of the season a decision is made to fish or not and whether to leave the ground (due to catch meeting the quota or if catch rates are sufficiently poor). If fishing occurs then the catch is determined according to the level of effort, catchability, vulnerability and abundance of fish. From these fish, some may be tagged and released and there may also be tagged fish recaptured amongst the catch. Catch rates are recorded and if sufficiently high over a period of time, there may be an increase in quota and an increase in the daily fishing frequency. However, if catch rates are consistently poor over a number of fishing days (8 days), then the vessel may depart the region on economic grounds. However, a high catch rate during the last 2 days may keep the vessel fishing. These 'leaving triggers' were formulated with the assistance of skippers from the vessels and had a major influence on simulation results.

7.1.5 The assessment model

The tag-recapture assessment model used in the MSE is exactly the same as that used in the annual assessment of Macquarie Island toothfish (the LDS Model of Section 5). The assessment model is a dynamic tag-recapture based model that includes population models of tagged and un-tagged fish. Estimates of pre-tagging available biomass, annual net recruitment and current available biomass are produced. Catch rate analyses are also performed. The current assessments are based on resource models of a single stock. Assessment models of two stocks are under development (Appendix C).

7.1.6 Decision rules

Three different decision rules for setting the annual TAC were tested in MSE analyses. Decision rule (ii) resembles that which is currently applied in the Macquarie Island toothfish fishery. These decision rules were:

(i) A fixed TAC set as a fraction, a, of the estimated initial available biomass B_o from an assessment,

$$TAC = aB_0 \text{, with triggers}$$

(*ii*) An annual TAC set as a fraction, *b*, of the estimated available biomass from the most recent assessment, $B_{current}$,

$$TAC = bB_{current}\text{, with triggers}$$

(*iii*) User defined fixed TAC. This allows special cases to be considered such as having no set TAC, so the fishery is driven by economic factors only, or having a zero catch to examine population recovery rates.

The TACs defined by the equations in (i) and (ii) are set in order to achieve sustainability criteria for the resident stock alone. However, the TAC is allowed to trigger up within the fishing season if the transient stock is thought to have been encountered. This is achieved by recording the daily catch rate of fish and if this exceeded 10 t/km^2 over 3 consecutive fishing days then the TAC is doubled. If the catch rates fall below this threshold level then the TAC reverts to the lower value, or if more catch than this value has been taken, then the fishery closes.

7.1.7 Results and discussion

Several management strategies were considered and applied across the various resource scenarios. The management strategies included: varying the annual number of fish tagged, altering assessment models, fixed TAC decision rules, feedback decision rules, differing the proportional exploitation parameters (*a, b*), having no TAC, and no fishing (to measure population recovery).

Evaluation of management strategies for the Macquarie Island toothfish fishery suggest that current fishing operations have had a small impact on the resource, and in particular on the spawning stock biomass (Figure 7.2). In addition, economic constraints, which force operators to abandon fishing during unfavourable catch conditions, can and have acted to conserve the resource (the leaving triggers). Tagging effort played an important role in decreasing the level of uncertainty in estimates of available biomass. Reducing tagging effort increased the likelihood that a tag-based assessment could not be performed and estimates of available biomass showed larger inter-annual fluctuations, producing greater variation in expected annual catch.

The feedback management strategies with $b = 0.1$ performed reasonably well across the considered scenarios (Figs 7.2 and 7.3). The TAC decision rules were generally able to protect the resident stock (through the lower TAC being set at the resident stock's sustainable level) and harvest the transient stock when it appeared (through an increase in quota via the TAC triggers). While it is encouraging that this management strategy outperformed the others, at least in terms of conservation benefits, it did falter (along with the other policies) in one key circumstance. Namely, if effort is markedly increased and there are two stocks present, resident stock spawning biomass may reduce to a larger extent than may be acceptable (Figure 7.3(c)). Under this scenario, fishing continues through periods of poor catch when the operators would normally have chosen to leave. The presence of transient fish leads to increased estimates of annual available biomass and hence TACs. The increased effort allows more of the TAC to be caught (it was rarely met otherwise) and subsequently more of the resident population is taken as part of the annual catch. To overcome this problem stock assessment models that explicitly model the resident and transient stocks are being developed.

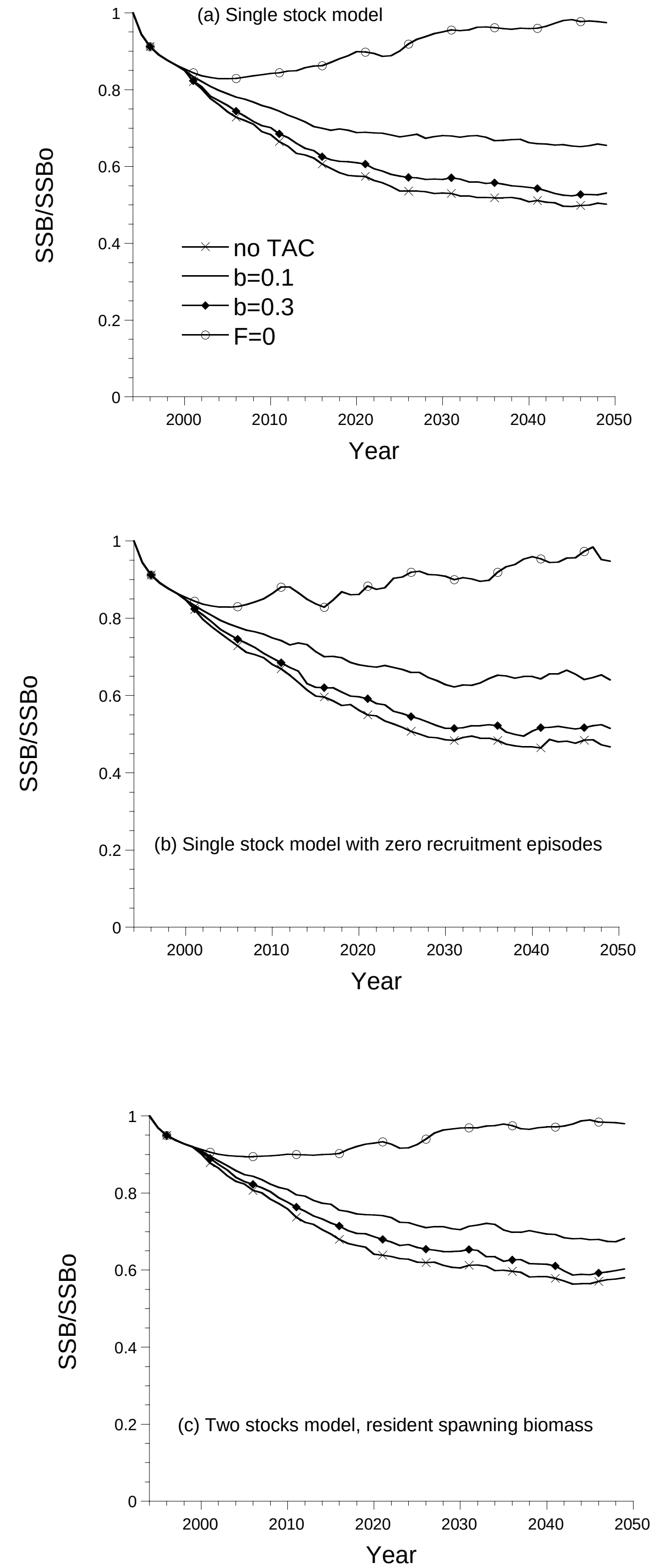


Figure 7.2. Time series of the median spawning stock biomass (SSB) relative to initial SSB for 4 management strategies and 3 resource scenarios. The No TAC management strategy sets no quota, so fishing operations are limited only by economic constraints (such as poor catch rates). The two b feedback management strategies set the annual TAC as the fraction b multiplied by the current estimate of available biomass. TAC triggers that can increase the quota within a season are allowed. The F=0 management strategy sets the catch to zero in all future projection years.

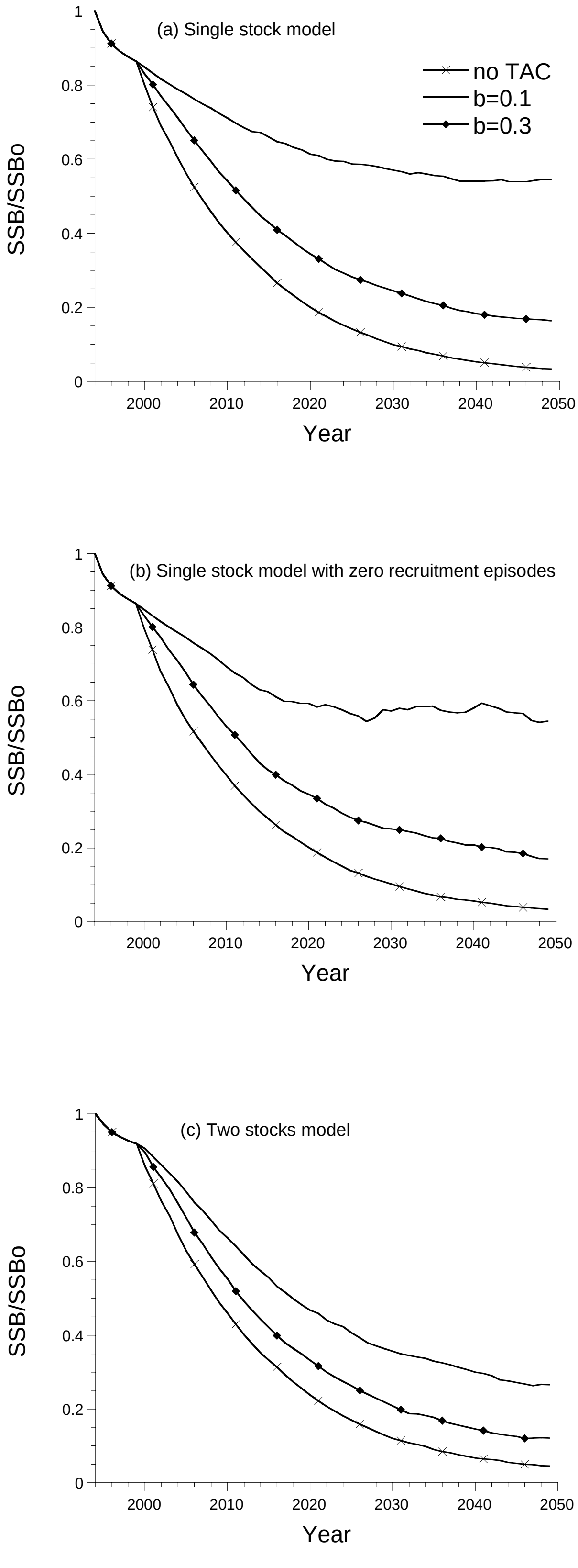


Figure 7.3. Time series of the median spawning stock biomass (SSB) relative to initial SSB for 3 management strategies and 3 resource scenarios where there are no economic or effort constraints, i.e. the vessel does not depart the grounds because of poor catch rates or season length. Note that the applied management strategies do not react to signals, such as declines in catch rates, which may indicate a need for a change in the TAC setting process.

The current stock assessment is based on a model of a single stock, and if two stocks are present it will provide an estimate of the combined biomass. However, estimates of resident biomass alone are required for depletion estimates and to set the annual TAC. As the assessment model cannot distinguish between resident and transient stocks, the pre-fishing resident available biomass level is taken to be the available biomass of the second season. This is because it is assumed that no transient fish appeared in this season. Once development, the two stocks assessment model will be able to estimate the abundance of resident and transient fish in the first season, overcoming the need to use an estimate of initial resident biomass from the second season. The two stocks assessment model will be able to track fluctuations in the resident biomass for all years, and also provide estimates of the magnitude of the transient fish when they appear (Appendix C). Figures 7.4 and 7.5 show that if the annual resident stock available biomass is known with certainty (perfect information) and the annual TAC is set to a fraction of this (e.g. b = 0.1), then the population of both resident and transient fish can be harvested in a sustainable manner for the resident stock. The question is then how to develop an assessment model that gives reasonable estimates of resident biomass.

An evaluation of management strategies showed that a range of policies, with appropriately chosen parameters, can satisfy sustainability criteria and maintain catch levels under most of the scenarios considered. Some management strategies considered can dramatically impact the stock even when effort is limited to the current levels in the fishery. The inability to catch the full TAC appears to be the controlling factor in the current fishery indicating that the current rules for establishing TAC's need to be refined so that they are an effective management tool should they ever be reached routinely.

There remain several areas of uncertainty to be explored and improvements can be made within the model structure, dynamics (biological and fishery) and management models. For example, key components of the model rely upon measures of catch rates in order to trigger either management responses or operational responses. As such, the modelling of catch rates is critical and should be considered carefully in future work. The models of toothfish population dynamics also require some consideration, as some management strategies are sensitive to resource model assumptions (noting however that an ideal management strategy should be robust to resource model choice). The software developed provides the opportunity to expand the resource hypotheses and management alternatives in a relatively expedient manner. For example models that consider alternative gears (e.g. longlining), effort regimes (e.g. extended seasons, multiple vessels) and resource hypotheses (e.g. MPAs, metapopulations) could be explored.

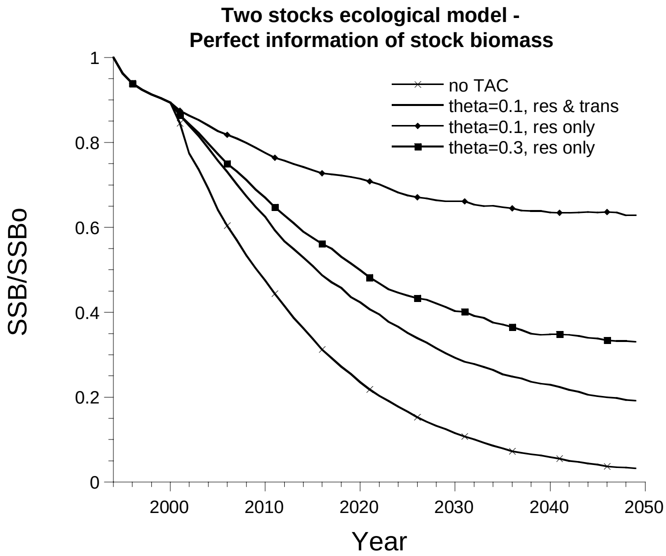


Figure 7.4. The time series of the median spawning stock biomass (SSB) relative to initial SSB for 4 management strategies and where operators are not constrained by season length nor economics that would otherwise lead them to depart the fishery. The population is composed of two stocks: a resident population and a transient population (spawning stock refers to the resident population). Theta gives the proportion of available biomass that is set to the current year's TAC. The ‘res only’ models use the available biomass of the resident population only to set the TAC. The ‘res & trans’ model combines the available biomass of both resident and transient fish when setting the TAC. Perfect information of biomass is assumed. Note that the applied management strategies do not react to signals, such as declines in catch rates, which may indicate a need for a change in the TAC setting process. Note that theta = *b* in the context of Section 7.1.

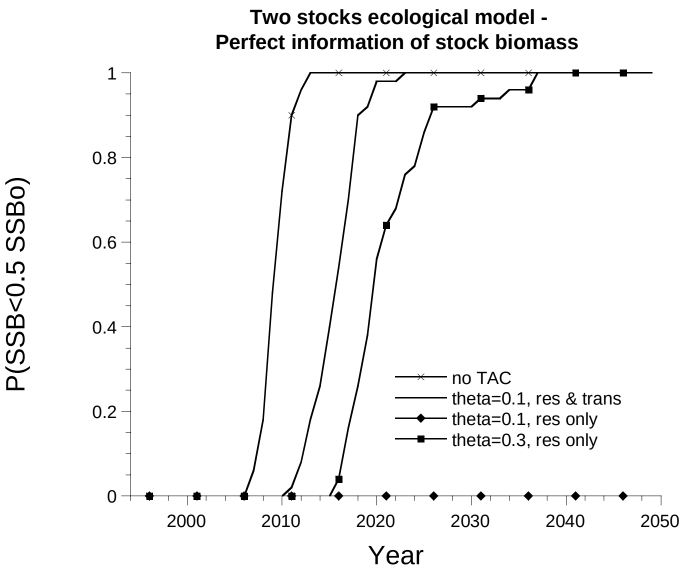


Figure 7.5. The time series of the probability that the spawning stock biomass is less than 50% of initial levels for 4 management strategies. Operators are not constrained by season length nor economics that would otherwise lead them to depart the fishery. The population is composed of two stocks: a resident population and a transient population (spawning stock refers to the resident population).

7.2 Bayesian Belief Networks[7]

7.2.1 Introduction

In this section methods are developed that demonstrate how knowledge related to biological hypotheses and observations can be brought into fisheries management modeling in a relatively simple way. Bayesian belief network (BBN) methodology is introduced and applied to the fishery for Patagonian toothfish at Macquarie Island. The belief network model includes biological observations from genetics, tagging experiments and fatty acid compositions. This section demonstrates that, in addition to catch limits as management tools, economic tools focusing on the behaviour of fishers can be effective in fisheries planning, especially in increasing the robustness of management systems. Moreover, changes within the utility function can reveal potential conflicts between various interest groups.

The precautionary approach uses the uncertainty of stock estimates as criteria to the applied exploitation level: the more uncertain information is, the lower the exploitation rate. In this context, the quality of knowledge is directly linked to management actions and a lower expected catch is the insurance fee paid when fishing a stock with highly uncertain status. This creates an incentive to improve the quality of information, which can then lead to higher exploitation. However, an alternative approach is to change the management system so that it can withstand current assessment errors, i.e. to create a more robust management system. In this case, actions are not made to improve the information, they are used directly to improve the system.

The planning of management systems is not easy. A well-managed system should not be sensitive to the assessment/monitoring information on a tactical level (incorrect estimates on a yearly level do not lead to catastrophes). Likewise, the system should not be sensitive to new causal relationships found by basic research, i.e. it should be robust to structural uncertainty, which is related to model selection when simulations are carried out. A good management system should also take into account the conflicting objectives of various interest groups and be robust to changes in objectives over time. This is especially important in cases where decisions are long-term, e.g. the establishment of areas closed to fishing.

Ideally, a management system should be robust to uncertainties in the data, models and objectives. We suggest that the value-of-information (Clemen, 1996) could be a useful metric when evaluating management options. If the value-of-information is low both on a tactical and strategic level (changing a decision is expected to yield low returns), there is little basis for change. We also suggest that the value-of-control (the gain obtained from using additional management tools) is an even more important tool in the evaluation and planning process. Thus, value-of-information is related to improving decision-making behaviour through better information, whereas value-of-control is related to finding new ways of manipulating the system.

This section applies the theory of Bayesian Belief Networks to the toothfish fishery of Macquarie Island (Kuikka et al., 2001). A major management problem in this fishery relates to the existence or otherwise of transient fish that occasionally increases the local biomass, and how uncertainty about their existence should be taken into account in a TAC policy. Control is based on the relationship between the quota and biomass of fish on the fishing grounds, and if available biomass is very variable, the control effect of the TAC is variable as well. If biomass is high due to the appearance of transient fish, then the TAC may be too restrictive. On the other

[7] Text and figures adapted from Kuikka et al. (2001).

hand, if transient fish are included when the available biomass is estimated to set the TAC, the quota might be too high for the resident stock if the transients have subsequently disappeared from the fishing grounds.

This section has the following objectives:

i. to develop a methodology that allows simple biological observations to be included in management analysis
ii. to evaluate how the tools of decision analysis can be used when planning fisheries management, especially when improving the self control of the system, and
iii. to show that subjective evaluation of probabilities for decision analysis modelling might be useful when planning future research and decision-making.

7.2.2 Methods

Observations of highly variable catch rates (CPUE) and variability in catch length-frequency distributions (unpublished data) has led to the hypothesis that a transient population of toothfish exists in the vicinity of Macquarie Island. Uncertainty related to the local stock dynamics is currently a major issue both for managers and for fishers. From the managers' point of view, highly variable and unpredictable available biomass decreases the usefulness of total allowable catches (TACs) as a management tool. This is because it increases the likelihood that the TAC and biomass do not match (the TAC is either too high or too low compared to the biomass). From the fishers' point of view, the highly unpredictable CPUE is a major source of economic uncertainty, as catch rates are an indicator of economic profitability.

It would be beneficial if scientific analysis could estimate the probability that there are transient fish in the area. Our approach to this question creates a belief network model that links genetic, fatty acid and tagging observations to biological hypotheses about the stock dynamics. This additional information can be used to make inferences about the probability that transient fish occur on the fishing grounds. The existence or otherwise of transient fish was modelled within a Monte Carlo simulation framework and the probabilistic outcomes of these scenarios were then used as input values for a decision analysis model. As new observations can update the probability that transient fish exist, we can analyse the role of biological knowledge in a management context.

The structure of the analysis is as follows:

1) In the first phase, we used the simulation results of the model presented in Tuck et al. (2001b) and Section 7.1. The simulation models include an operating model of the fish and fishery, a sampling model, an assessment model and a management model.
2) In the second phase, we constructed a simple biological ("diagnostic") model by belief network methodology. Diagnostic in the sense that it combines uncertain observations to alternative hypothesis concerning the "state of nature", in particular the existence of transient fish in the fishery.
3) In the third phase, we constructed a decision model based on the probabilistic results of the simulation model and linked the diagnostic model and decision model in order to demonstrate how much basic biological knowledge matters in a decision making context. In the decision model we use variants of the utility function (a function which describes the managers' objectives and is used to rank decision alternatives, in our case a combination of

yield and effects on spawning stock biomass) to rank the decision alternatives and to evaluate the importance of different objectives.

There are several published applications of BBNs to fisheries and environmental systems (Varis et al., 1990; Varis and Kuikka, 1997; Hilden, 1997; Kuikka et al., 1999; Hammond and O'Brien, 2001). Moreover, Jensen (1996), Almond (1995) and van der Graag (1996) give clear presentations on the use of belief networks.

The idea behind BBN is fairly simple. They mimic human inference: the human mind connects variables by means of logic and greater weight is given to better knowledge. The amount (in a qualitative sense) of knowledge is described by probability distributions and the model is used for uncertain reasoning or probabilistic reasoning. If something is unknown (i.e. no information), that part of the model should not be reflected in other parts of the model. Knowledge is 'collected' from different parts of the model and the structure of the entire model is uncertain, not only the parameters. According to Kuikka and Varis (1996), the close relationship between human thinking and belief networks can be seen in the process where knowledge is obtained from experts for the models (called elicitation). It appears easy for experts to include their logic and other knowledge in the belief networks.

The model consists of nodes, i.e. variables, that have two or more possible states, and the state has a given or calculated probability. A parent node is a node with leading arrows (indicating a dependency), and a child node is a node with incoming arrows. The strength of inter-dependency of nodes is described by conditional probability values (e.g. Jensen, 1996) or link values (Pearl, 1988). The weaker the dependency between the nodes, the less a parent node can introduce information through the child node. If a parent node is a decision node (its value can be chosen and implemented in practise), the degree of controllability can be described with conditional probabilities (a high degree of determinism leads to low uncertainty in the conditional probability distribution).

If a model does not include any observations, the information content is dependent on the given prior probabilities. Once new information (through observation) is introduced, the rest of the network is updated by the information content of the observation, and by the degree of dependency between the variables. Put simply, if several variables are correlated, then knowing one variable should update the knowledge about the state of the other variables.

Figure 7.6 shows a typical example of a simple resource management model. The "State of nature" (e.g. the number of fish in the stocks) is unknown. The stock assessment can give some useful information about this, but only on a probabilistic basis. This information is available at the time decisions are made, and achieving the objective depends on how well the TAC matches with the real size of the stock (fulfilling the objective is dependent both on the state of nature, and on the decision made).

Variables of the influence diagram

The elements of the model are described in Figure 7.7 and the model is described in Figure 7.8. Note that the first part of the results includes only one decision variable: the management strategy. The results concerning the value-of-control include two decision variables (management strategy and assumed economic control).

Variables of the biological (diagnostic) model

Tagging distribution. This observation node has two outcomes (Table 7.1): mixed (adults of the two fishing areas mix between the two fishing grounds) or separate (adults do not mix). As tag returns are only from grounds surrounding Macquarie Island, this observation can not be linked to the transient fish node. It only has a role with respect to the locality hypothesis (below). However, the results show that it has information content with respect to transient fish, depending on what is observed.

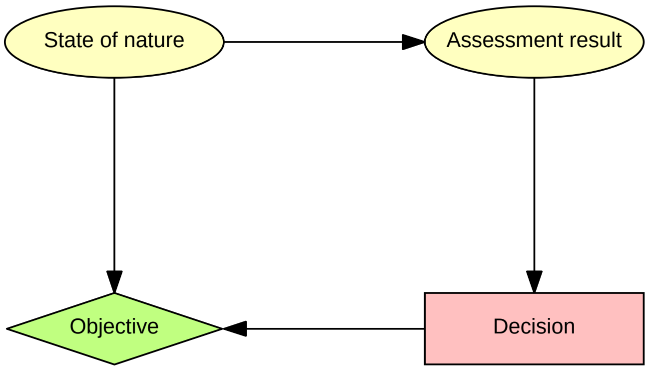


Figure 7.6. Simplified structure of a typical resource management model. Information flows through the assessment to the management decision. Results are then dependent on what was done (e.g. TAC decision) and the real state of nature (e.g. size of the stock).

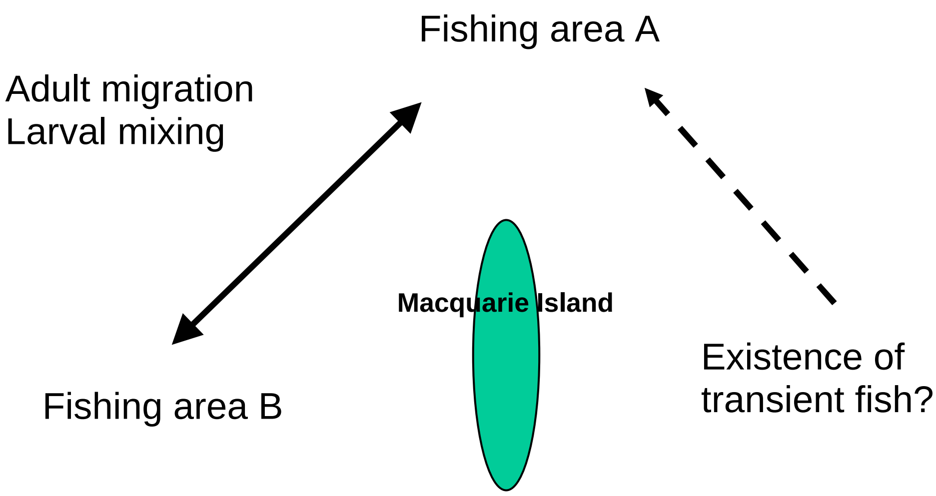


Figure 7.7. A schematic description of the elements of the biological input to the belief network models. Observations from the fishing areas are: genetic samples, tagging data, fatty acid composition. See text for details.

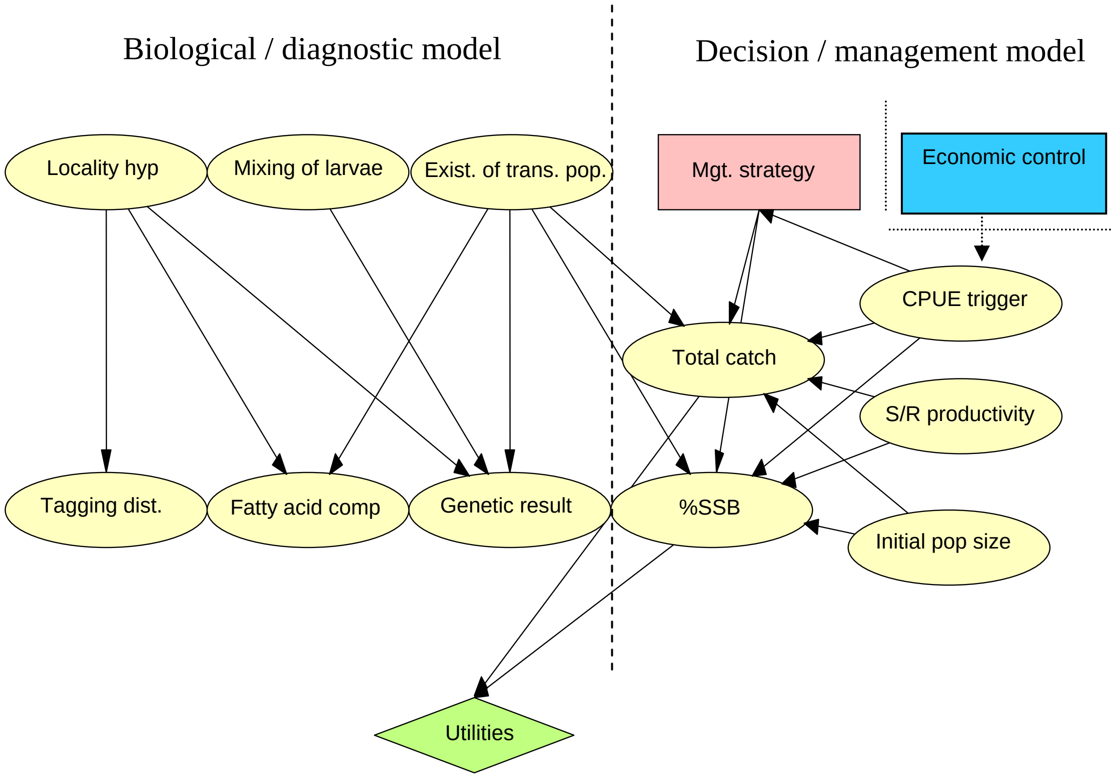


Figure 7.8. The structure of the influence diagram model. The 6 nodes (variables) on the left represent the biological ("diagnostic") part of the model and the rest of the model represents the decision model. Economic control is an optional control.

Table 7.1. Conditional probabilities for the tagging data. Values are conditional on the locality hypothesis. Tot. sep. = adults do not migrate between the two fishing grounds. Mixed = adults are mixed between the two fishing grounds.

Locality hypothesis	Tot. sep.	Mixed
Mixed	0.05	0.95
Separate	0.95	0.05

Table 7.2. Conditional probabilities for the statistical test of fatty acid data (is there a difference between the two fishing areas). Values are conditional on the existence or otherwise of transient fish and the locality hypothesis. Diff. obs = genetic test gives a difference, No diff obs. = genetic test does not give a difference.

	Transient, gen. diff. population		Transient, gen. sim. population		No transient population	
Variable	Tot. sep.	Mixed	Tot. sep.	Mixed	Tot. sep.	Mixed
Diff. Obs.	0.79	0.75	0.79	0.75	0.37	0.35
No diff. Obs.	0.21	0.25	0.21	0.25	0.63	0.65

Locality. This hypothesis node includes two outcomes: fish are local or mixed. By local we mean that adult fish do not mix between the two major fishing grounds. Mixed is the opposite of the local assumption. As the prior probabilities show (Table 7.2), there was high initial belief that toothfish are an active predator that mixes on a local scale around Macquarie Island.

Mixing of larvae. This hypothesis node has two assumptions about the larval biology of the local fish: they are either mixed (thereby removing the chance that a genetic difference exists between the local populations of toothfish) or that they are separate, which means that the larvae of the two fishing grounds migrate back to the area where their genetic components originate. This hypothesis is required to consider the chance that genetic differences are based on local fish alone. This may then decrease the statistical power of genetic results with respect to the existence of transient fish. As indicated by the priors, it was considered almost certain that toothfish larvae mix randomly on the scale considered.

Transient fish. This hypothesis node has three possible outcomes: No transient fish (the available stock consists of local fish only), Genetically different fish (transient fish exist and are genetically different from the local ones) and Genetically similar fish (transient fish exist and are genetically similar to the local ones). This is the main focus of the diagnostic side of the model and creates a link to the decision analysis part of the model. The high prior for the existence of genetically similar transient fish is based on the fact that variability in CPUE in the very beginning of the fishery suggested that there might be transient fish in the area.

Fatty acid composition. This observation node has two outcomes: different (the fatty acid composition is statistically different between the two areas) or similar (no difference). The probability distributions created by the prior probabilities of the model (Table 7.2) depend on the power of the test, as well as on the priors given for “transient” and “locality” nodes.

Genetic results. This observation node has two outcomes (Table 7.3): different (the genetic backgrounds of the fish are statistically different between the two areas) or similar (no difference). Early genetic tests suggested there was a difference between local populations, however more recent studies have concluded there is no difference. The prior for the observation depends on the power of the genetic test, as well as on the priors given for “transient”, “locality” and “mixing of larvae” nodes.

Table 7.3. The conditional probabilities for the result of genetic tests. Values are conditional on hypothesis on larvae mixing, existence of transient fish and locality. Diff. obs = Genetic test gives a difference, No diff obs. = genetic test does not give a difference.

Mixing of larvae - hypothesis	Larvae are mixed					
Transient fish hypothesis	Transient, gen. diff. population		Transient, gen. sim. population		No transient population	
Locality hypothesis	Tot. sep.	Mixed	Tot. sep.	Mixed	Tot. sep.	Mixed
Diff. Obs.	0.8	0.8	0.02	0.02	0.02	0.02
No diff. Obs.	0.2	0.2	0.98	0.98	0.98	0.98

Mixing of larvae - hypothesis	Larvae are separate					
Transient fish hypothesis	Transient, gen. diff. Population		Transient, gen. sim. population		No transient population	
Locality hypothesis	Tot. sep.	Mixed	Tot. sep.	Mixed	Tot. sep.	Mixed
Diff. Obs.	0.88	0.8	0.08	0.02	0.08	0.02
No diff. Obs.	0.12	0.2	0.92	0.98	0.92	0.98

Variables of the decision model

The following items are explained in more detail in Section 7.1 and Tuck et al. (2001a, 2001b). Here, an overview is given that allows an understanding of the general behaviour of the influence diagram.

Management strategy. This decision node includes 5 different management options that could be applied in the simulation model:

1) **Fixed TAC.** This is based loosely on the CCAMLR precautionary strategy, where a certain proportion (10% in this case) of the estimated available initial biomass is used as the fixed TAC for all future years (Anon. 1994).

2) **No TAC**. No restrictive TAC. Cessation of fishing is based only on the behaviour of the fleet, i.e. the vessel leaves the fishery when the CPUE is economically too low.

3) **Tag 500, θ=0.1.** 500 fish are tagged every year, an assessment is made and the following year's TAC is 10% of the estimated current available biomass.

4) **Perf, θ=0.1**. Perfect information is assumed (the exact biomass is known at the time the TAC decision is made) and the following year's TAC is 10% of the current year's available biomass.

5) **Perf, θ=0.3**. Perfect information assumed and the following year's TAC is 30% of the current year's available biomass.

Total catch. This node is the mean annual catch over all projected years. This variable describes the economic interest of fishing.

Leaving trigger. The fishery model includes a trigger level such that if the mean CPUE is not sufficient over the last 8 consecutive days of fishing, the vessel departs the fishery. However, a high catch rate during the last 2 days may keep the vessel fishing. Three discrete values were chosen as the CPUE trigger values (3, 5 or 7 tonnes per km^2 per day).

Stock-recruitment. This node includes two options (0.5 and 0.75) for the steepness value of the Beverton and Holt stock-recruitment (S/R) function. It is an important part of the reproductive capacity of the stock, and initially it was assumed to be a major source of biological uncertainty.

%SSB. This node includes the probability distribution for the relationship of spawning stock biomass (SSB) in the last year of the simulation horizon divided by the initial SSB. This variable describes both the recruitment risk for the target species, as well as the potential of the fishery to impact other parts of the food web.

Initial population size. This node has two possible outcomes: 20% and 100% of the current point estimate. The size of the available population before fishing began is used to estimate the S/R parameters, as well as criteria describing how much the spawning stock decreases due to fishing. This variable is uncertain, even though existing studies show that assessments have been fairly stable for different assumptions. However, in the assessment it is assumed that the survival of tagged fish is the same as that of un-tagged fish in the population. Tagging mortality could lead to an overestimation of the population abundance and therefore the initial available population size may be overestimated. Also, the possible disappearance of tagged transient fish might lead to an underestimation of the population. As such, the probability of the initial population being only 20% of the best estimate was assumed to be 0.2 and the probability that the initial population is the best estimate was assumed to be 0.8.

Elicitation of probabilities

Both the conditional probabilities applied in the diagnostic model (shown by incoming arrows in the model) as well as the prior probabilities of the unconditioned variables (variables with leading arrows only) were asked of experts from each field. There were two genetic experts, two fatty acid experts and three experts gave their opinions on the (prior) variables: Locality hypothesis, Mixing of Larvae, Existence of transient population. The genetic and fatty acid experts gave only one set of agreed probabilities, whereas the prior probabilities were given in two sets and the final probabilities were combinations of these probabilities.

Problem description and logic of the model

The conditional probabilities used in the left side of the model (i.e the biological or diagnostic model; Figure 7.8) are given in Tables 7.1 – 7.3. The information content of the diagnostic model is based on these. In the following text, we describe the logic of the model, as well as the logic of the probabilities.

We start by creating the required hypotheses and by linking observations to these hypotheses. There are two separate fishing grounds around Macquarie Island, and possibly a transient fish population outside of the area. Data sets exist that may include useful information on this inference. These are: genetic samples, fatty acid samples, and tagging data sets from the fishing grounds.

Genetic observations are potentially very useful: if the transient fish can be shown to be genetically different from local fish, this effectively shows that there must be another population in the area. This has important implications for management of the stocks (both resident and transient). However, it is theoretically possible that if the adult fish do not migrate between the two fishing areas, a statistically significant difference in genetic composition could be obtained on the basis of the local fish alone. Therefore, the "Locality" hypothesis is also needed and tagging data is informative for this hypothesis. If larvae are well-mixed and return randomly to the fishing grounds, a genetic difference between the areas could not exist even though adult fish were separated. Therefore, hypotheses about larval mixing were included in the analysis, even though no data exist to update the probabilities for this hypothesis. Even prior information such as this, where there is no ability to update the probabilities, can be useful. If larvae are well-mixed (higher part of Table 7.3), experts stated that the chance of obtaining a statistical difference between the fishing grounds if there is no transient fish was 0.02. If larvae and adults are separate, this probability was assumed to increase to 0.08 (the effect could be from local fish alone). The power of the genetic study is assumed to be high: experts stated that they assumed a probability of 0.8 of seeing a genetic difference, if there is one. Clearly, a statistical test showing a difference would be a very informative observation here.

The tagging data can not directly update information about transient fish, because fish are tagged only on the two fishing grounds and transient fish have most likely not been tagged. However, as mentioned, tagging data can provide information about the site specificity of the fish, which can be linked to the genetic observations and thereby illustrate the potential for a genetic difference.

Fatty acid data have a different role than the genetic observations. Unlike a genetic "label" that stays with the fish, if transient fish are eating the same prey as the local fish, a difference in fatty acids will disappear in about 5 months. However, if sampling occurs soon after the transients arrive, a difference in fatty acids could be observed even though the transient fish were not genetically different from the local ones. There is a risk that this difference is due to the different diet of the fish on the two fishing grounds, as Aurora Trough and the Northern Valleys may experience quite different current driven prey availability. This also has implications regarding the degree of movement between the local grounds. All of these alternative explanations contribute to the probability of getting a significant statistical result without the transient fish. Experts evaluated that there was a probability of 0.75 of observing a difference if transient fish existed and local fish were mixed between the two fishing areas (Table 7.2). The existence of locally separate fish increases this probability to 0.79. However, if there were no transient fish in the area, a difference would be fairly probable, i.e. 0.35 for mixed adults (or 0.37 for locally separated adults). The observation of a difference in fatty acid composition is not very informative about transient fish as there is a high risk of getting this result based only on local environmental differences (like the diet of fish on a local scale).

Utility function

In decision analysis applications, a utility function describes the interests or objectives of the decision-maker. It can be considered to be a systematic tool to analyse the ranking of the management options according to their different assumptions. A “basic run” utility function to describe the fisheries’ management principles was found through discussions with managers responsible for the Macquarie Island fishery, in addition to our own experience. In formulating a utility function one should ask, for example, what magnitude of assured catch is equivalent in utility to a manager as a game where 500 000 tonnes are caught with probability 0.5, and zero catch with probability 0.5. If the answer to this question is less than 250 000 tonnes the person is risk averse, and if more than 250 000 tonnes, risk seeking.

Table 7.4. The conservation oriented utility function. Note that the discrete classes are not equal. %SSB refers to the percent reduction in the local stock's spawning stock biomass.

	Yield (total yield)												
%SSB	0-20	20-50	50-100	100-150	150-200	200-250	250-300	300-350	350-400	400-450	450-500	500-550-	> 550
100	0.71	0.74	0.77	0.82	0.85	0.88	0.90	0.93	0.94	0.96	0.97	0.98	1.00
90	0.68	0.71	0.75	0.79	0.83	0.86	0.88	0.90	0.92	0.94	0.95	0.96	0.97
80	0.66	0.68	0.72	0.76	0.80	0.83	0.85	0.87	0.89	0.91	0.92	0.93	0.94
70	0.62	0.65	0.68	0.73	0.76	0.79	0.82	0.84	0.85	0.87	0.88	0.89	0.90
60	0.57	0.60	0.64	0.68	0.72	0.75	0.77	0.79	0.81	0.83	0.84	0.85	0.86
50	0.29	0.32	0.36	0.40	0.44	0.46	0.49	0.51	0.53	0.54	0.56	0.57	0.58
40	0.17	0.20	0.24	0.28	0.31	0.34	0.37	0.39	0.41	0.42	0.44	0.45	0.46
30	0.09	0.12	0.16	0.20	0.23	0.26	0.29	0.31	0.33	0.34	0.36	0.37	0.38
20	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

The applied utility function is given in Table 7.4. The basic utility function is risk averse. An additional unit in yield and %SSB is assumed to be more valuable when their values are low, i.e. a manager is more concerned about an increase in %SSB from 40-50% than from 90-100%. It was assumed that it is not acceptable for the spawning stock to reduce below 20% of the virgin spawning stock biomass. "Risk neutral" and "Business-oriented" utility functions were considered in Kuikka et al. (2001). The "risk neutral" utility function assumes that an additional unit of spawning stock biomass or yield has the same utility on low levels as high levels. The "Business orientated" takes into account that Macquarie Island is a long distance from ports and steaming costs are high, therefore good catches may be required in order to cover expenses. In this function, %SSB has no role, even though a large biomass would produce large catches.

7.2.3 Results

Biological (diagnostic) model

In the diagnostic part of the model (Fig 7.7; left side) the main interest is to analyse how the results of the statistical tests change other probabilities, especially the probability of the existence of transient fish. The results of making optional single observations are given in Table 7.5. The first number column on the left gives the probabilities without biological observations included in the model. These are prior probabilities, i.e. they are determined from knowledge before any observations are made. Experts gave a fairly high probability for the existence of the transient fish.

The tabled conditional probabilities include the power of the statistical tests, i.e. the probability of seeing a difference if it really exists. For example, the probability of observing a genetic difference is 0.06. Although low, it is higher than the probability of the existence of genetically different transient fish, being 0.05. This difference is due to the fact that observations of a genetic difference could be based on local fish alone. The probability given for separate local fish (0.15) and larvae not mixing between grounds (0.05) includes enough information to increase the probability of having a genetic difference above 0.05.

In Table 7.5 we show by example how the diagnostic model reacts to new information. In each alternative (a) to (f), it is assumed that observations are obtained and these update knowledge in the rest of the network. The observations are assumed to be certain, i.e. the probability given for an alternative is always 1.

In alternative (a), a genetic difference is observed. This increases the probability for genetically different transient fish from 0.05 to 0.67. The opposite observation ((b), no difference) returns probabilities close to the prior situation, even though the probability of genetically different transient fish decreases. These large changes in probabilities are based on the relatively high power of genetic results.

In alternative (c), a difference in fatty acid composition is observed. This decreases the probability of “no transients” from 0.3 to 0.17. The opposite observation (d) increases the likelihood of no transient fish to 0.53. Observations from fatty acid analyses are not as informative as genetic results, as fatty acid results could be obtained by factors other than from transient fish.

Tagging data only substantially update the “locality hypothesis”. The conditional probabilities can not influence the transient fish probabilities as fish were only tagged on the two local grounds and no returns outside of Macquarie Island were made.

The lower part of Table 7.5 gives values of the utility function (conservative compromise between yield and biological risk) for each management alternative after an observation is made. The values range from 0.56 (No TAC, alternative (d)) to 0.7 ("Tag 500, θ=0.1", alternative (a)). Values of the utility function are directly related to the probability of the existence of transient fish (the higher this probability, the higher the utility) as catches will then increase without an increase in risk for local fish. These values demonstrate the effect of biological knowledge on management interests. In each column the relative differences in the utility functions are small, indicating that differences between management options is small. Note that the relative ranking of the decision alternatives stays the same, i.e. "Tag 500, θ=0.1" is always the best option, independent of the observations made.

Table 7.5. Effects of observations on the posterior probabilities of the diagnostic model. The 5 lowest lines include the value of the utility function for different management alternatives after the observations are made. Observation alternatives (a, b, …) are explained in the text. In each case, the alternative having probability 1.0 is assumed to be observed.

			Observation alternatives				
Type of node	Name of node	Outcomes	Basic (priors)	a	b	c	d
Hyp.	Locality hyp	Separate	0.15	0.16	0.15	0.16	0.14
		Mixed	0.85	0.84	0.85	0.84	0.86
Hyp.	Mixing of larvae	Mixed	0.95	0.94	0.95	0.95	0.95
		Separate	0.05	0.06	0.05	0.05	0.05
Hyp.	Existence of trans.	No trans	0.30	0.10	0.31	0.17	0.53
		Gen. diff.	0.05	0.67	0.01	0.06	0.03
		Gen. similar	0.65	0.22	0.68	0.77	0.43
Obs.	Tagging dist.	Mixed	0.82	0.81	0.82	0.81	0.83
		Separate	0.19	0.19	0.18	0.19	0.17
Obs.	Fatty acid compos.	Difference	0.64	0.71	0.63	1.00	0.00
		No diff.	0.36	0.29	0.37	0.00	1.00
Obs.	Genetic result	Difference	0.06	1.00	0.00	0.07	0.05
		No diff.	0.94	0.00	1.00	0.93	0.95
Utility	Utility function	Fixed TAC	0.64	0.68	0.64	0.67	0.59
		No TAC	0.60	0.64	0.60	0.63	0.56
		Tag 500, 0.1	0.66	0.70	0.66	0.69	0.62
		Perf, 0.1	0.65	0.68	0.64	0.67	0.60
		Perf, 0.3	0.61	0.66	0.61	0.64	0.57

As demonstrated in Table 7.6, the real value of the BBN is its ability to combine probabilistic information. Alternative (a) includes basic observations made during the analysis. These observations were:

a) that there is no genetic difference between the fishing grounds (Appleyard et al., unpublished),

b) that there was a difference in fatty acid composition between the fishing grounds (Wilson and Nichols, 2001), and

c) that the adult fish do not migrate between the two fishing grounds (Williams and Lamb, 2001).

Table 7.6. Effects of some observation combinations on the posterior probabilities of the biological (diagnostic) model and on the values of utility functions. a = observations made during the project.

			Observation alternatives			
Type of node	Name of node	Outcomes	a	b	c	d
Hyp.	Locality hyp	Separate	0.78	0.16	0.15	0.79
		Mixed	0.22	0.84	0.85	0.21
Hyp.	Mixing of larvae	Mixed	0.95	0.95	0.95	0.92
		Separate	0.05	0.05	0.05	0.08
Hyp.	Existence of trans.	No trans	0.18	0.18	0.31	0.05
		Gen. diff.	0.01	0.01	0.01	0.69
		Gen. similar	0.81	0.81	0.68	0.25
Obs.	Tagging dist.	Mixed	0.00	0.81	0.82	0.00
		Separate	1.00	0.19	0.18	1.00
Obs.	Fatty acid compos.	Difference	1.00	1.00	0.63	1.00
		No diff.	0.00	0.00	0.37	0.00
Obs.	Genetic result	Difference	0.00	0.00	0.00	1.00
		No diff.	1.00	1.00	1.00	0.00
Utility	Utility function	Fixed TAC	0.67	0.67	0.64	0.70
		No TAC	0.63	0.63	0.60	0.65
		Tag 500, 0.1	0.69	0.69	0.66	0.71
		Perf, 0.1	0.67	0.67	0.64	0.69
		Perf, 0.3	0.64	0.64	0.61	0.67

Together, these observations have reduced the probability of “no transient” from 0.3 to 0.18. This is because the observation of a fatty acid difference supports the hypothesis of transient fish. However, as this observation might be based on differences in the local environment, there is not a large change to this probability.

In alternative (b) the information content of the tagging observations is removed (i.e. the model behaves as though the tagging program had not existed). “Locality” probabilities change remarkably, but there is no effect on the probability of the existence of transient fish as it is assumed that no transients have been tagged. In alternative (c), removing fatty acid knowledge, in addition to tagging data, increases the chance that there have not been transient fish in the area.

In alternative (d), we return to knowledge when the development of this model began. Preliminary genetic results surprisingly suggested that there were genetic differences between the fish of the two fishing grounds around Macquarie Island (Reilly et al., 2001). As expected, this had a major effect on the probability of the existence of transient fish. However, the locality of adults does not create a genetic difference if the larvae were well-mixed and return randomly to the fishing grounds.

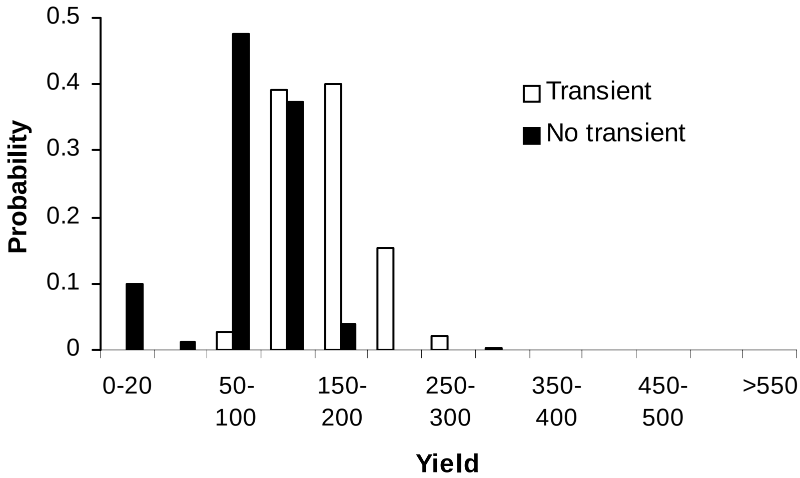
An essential advantage of using belief networks is that the effect of conflicts between information sources can be evaluated. Considering case (a) of Table 7.6, the genetic result does not support the existence of genetically distinct transient fish, and the probability of "No transient fish" is 0.18. In case (d), the genetic result supports the existence of genetically different transient fish, and the probability of "No transient fish" reduces to 0.05.

Behaviour of the decision part of the model

The biomass and yield predictions of the decision model (right side of Figure 5), including the probabilities estimated by the simulation model, are very sensitive to knowledge about the transient fish. In Figure 7.9, the effect of the transient fish hypotheses on the %SSB of local fish and on the total yield (resident and transient) is given. This hypothesis appears important: having transient fish in the area increases catches and decreases the biological risk for the local stock.

Figure 7.10 shows that observations seen during the project that supported the existence of transient fish have reduced the probability of a stock collapse (%SSB=0) in the resident population. The model is also sensitive to the probability distribution of initial population size. As Figure 7.10 demonstrates, the probability of stock collapse increases if the initial population is less than current best estimates indicate.

Surprisingly, the decision model is not that sensitive to the parameters of the stock-recruitment (S/R) function (Figure 7.11). This is based on the fact that the leaving trigger applied in the simulation model effectively controls the total biomass of the stock, keeping it on the stable part of the Beverton – Holt S/R function. This can be seen in Figure 7.12, where the probability distribution of the %SSB is given for three different CPUE trigger values. The lower the leaving trigger CPUE value, the lower the %SSB level (fishing continues on a lower biomass level).



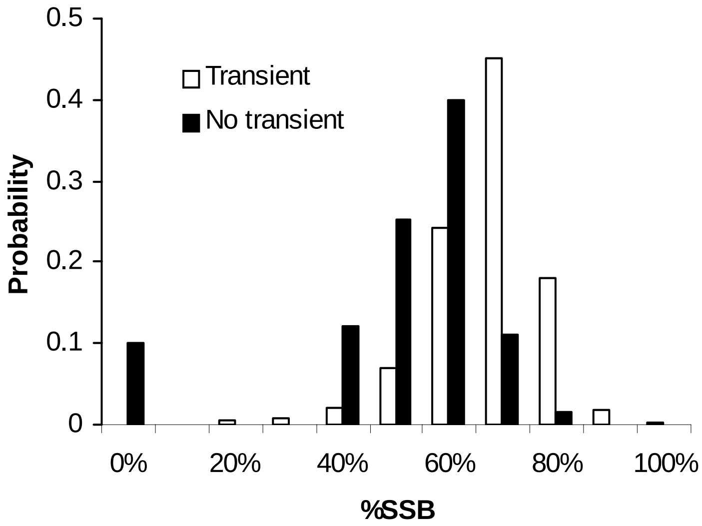


Figure 7.9. Probability distribution of %SSB (resident stock only) and total yield (resident and transient) assuming that transient fish either exist or do not exist. Other probabilities in the model are prior probabilities and a Fixed TAC management scheme has been applied.

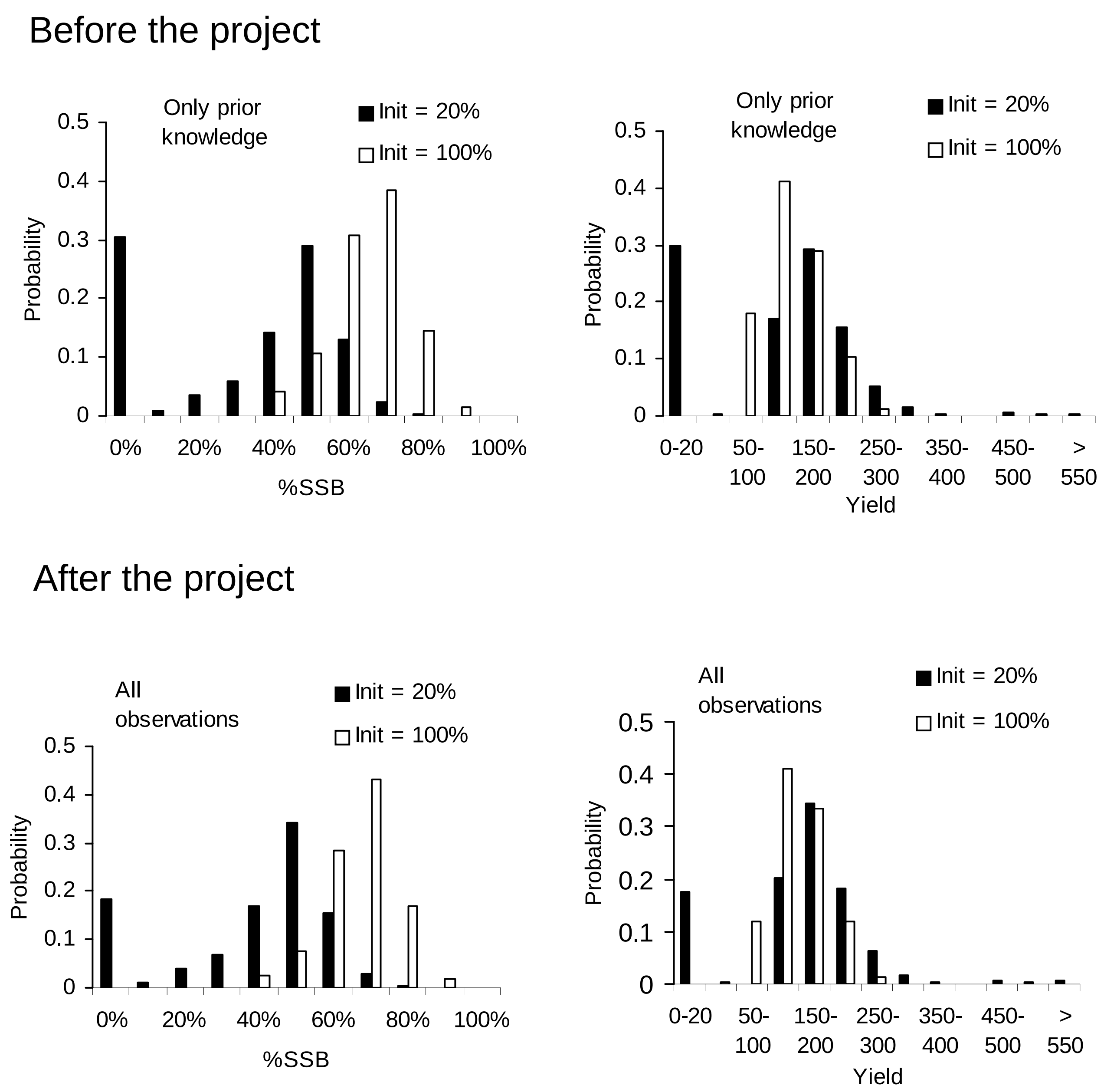


Figure 7.10. Probability distributions of the percentage of virgin spawning biomass (%SSB, resident stock only) and total yield for a Fixed TAC decision rule when information is based on prior probabilities only (upper figure), and when current observations (Table 7.3, alternative a) are used (lower figure). Probability distributions are given with two different assumptions about initial population size (20% and 100% of the point estimate).

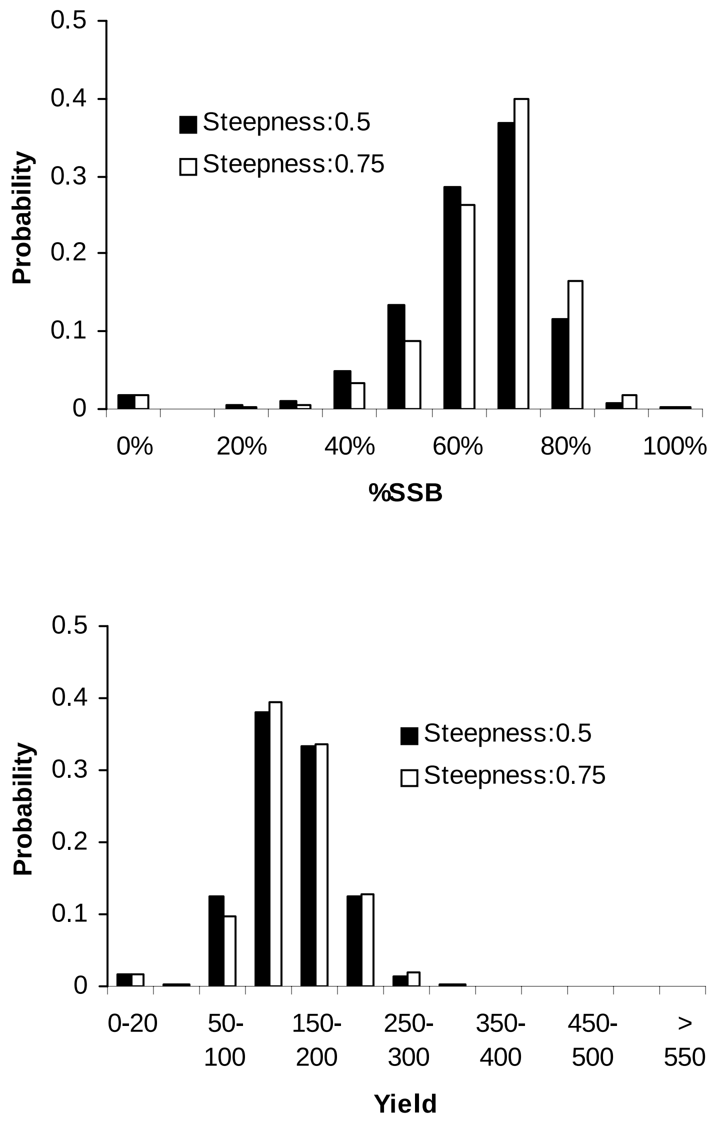


Figure 7.11. Probability distributions of %SSB (only resident stock, upper figure) and total yield (lower figure) when current observations and a Fixed TAC decision rule are applied. Probability distributions are given for two different assumptions about the steepness value of the Beverton - Holt S/R function.

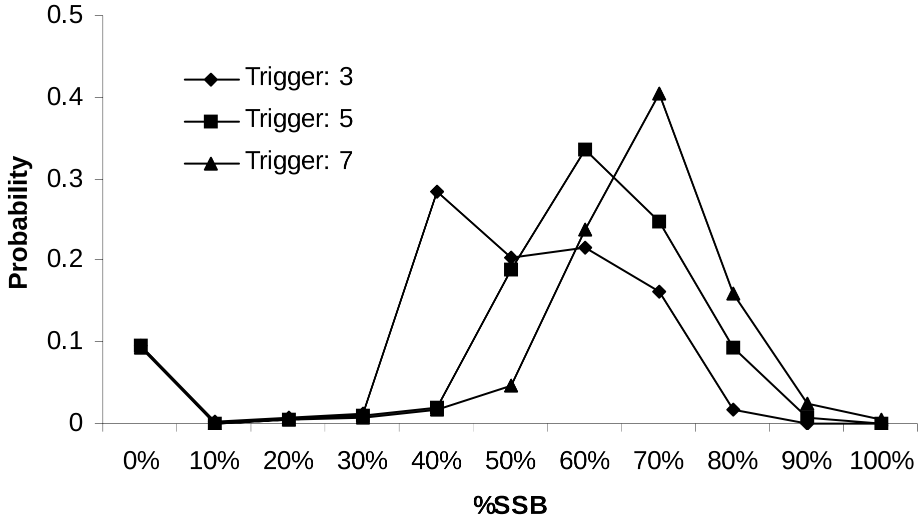


Figure 7.12. Probability distributions for a fixed TAC management rule when information in the diagnostic part of the model is based on current observations (Table 7.3, alternative a). Probability distributions are given for three different assumptions about the leaving trigger CPUE value (tonnes per km^2).

7.2.4 Discussion

The role of basic biology

Combining biological observations directly to the management model is a new one in fisheries management, even though these types of models are fairly common in medical sciences where the need to combine the information content of diagnostic inferences to decisions is obvious. Similar logic should be useful in fisheries management.

The diagnostic model could include more alternative hypothesis, which may explain other results. An example is water mass movements in the area (Goldsworthy et al., 2001). Water movements may have an affect on the diet of the toothfish, and consequently on their fatty acid composition. As Table 7.3 demonstrates, the assumption of larval mixing has an important role if a difference in genetic composition exists and the tagging results suggest that the populations are locally separated. This increases the probability that the genetic difference is based on local fish only. However, if it is known that larvae mix, there is a low probability that a genetic difference based on local fish alone would be observed. Therefore the relative role of the biological knowledge of the larvae only has a role when joined with certain other combinations of observations.

It is clear that the logic applied to the structure of the inference model is an essential role. There are certainly several alternative ways of constructing this type of model, and therefore each of them would give different results. These results are always subjective, and this is one reason why a Bayesian context fits the task well.

In the decision context (when using utility functions), the knowledge related to the biological observations in the diagnostic model were much more important than the S/R model parameters applied, demonstrating that biological knowledge might have a large impact on catch and biomass predictions. However, this is partly based on the use of CPUE triggers in the simulation model, which kept the spawning stock biomass at a level where there is no real risk of recruitment overfishing. The importance of the diagnostic model is based on the assumption and observations that potentially, if the transient fish hypothesis was true, the additional biomass is so high that it can remarkably increase yield without damaging the local stock spawning biomass. This was an important result derived from the application of the conservative utility function.

The task of prediction versus the task of ranking actions

The catch and biomass predictions were fairly sensitive to the probability associated with the transient fish hypotheses. This is expected, as transient fish dramatically increase the biomass available to the fishery. However, we demonstrated that the ranking of the decision alternatives (i.e. the decision rules of the management strategies) is not very sensitive to this knowledge. This is a strong feature of decision analysis: it may be difficult to predict exactly what will happen in the future, but there might be enough information to rank the decision alternatives. It is easier to recommend an action in this case, than to predict what will happen.

Value-of-information

It is clear that improved information offers opportunities to make choices that result in better outcomes. If information obtained at the time of decision making leads to a different action with a better outcome, the information has been beneficial (an outcome which is valued by the aid of the objective function). Better results (e.g. catch) of decisions are the basis for calculating the expected value-of-information.

The expected value-of-perfect-information (EVPI) describes the maximum price that should be paid for knowing the exact value of a variable at the time of making a decision, in contrast to having uncertain information about the variable. It shows which uncertainties matter and by how much. It is useful when deciding which variables to monitor.

EVPI is the expected value of the utility function in the model where the variables are known exactly minus the expected value of the utility function of the model where the values are not known exactly. Kuikka et al. (1999) estimated EVPI for mesh size decisions in the Baltic cod fishery. The value-of-information considered here was the difference between the utility of a perfect knowledge run (i.e. a run where estimates of available biomass through an assessment were assumed to be exact) and an assessment model run (i.e. a run including biases from the assessment). When these two values are compared, the difference shows the impact of uncertainty of the assessment results in terms of the biomass or yield.

The results concerning the value-of-information were somewhat surprising. In this case, we compared the values of decision alternative “Tag 500, θ=0.1” to the values of perfect information. The expectation was that by improving assessment estimates a better result from management should be obtained. However, in this case the stock was underestimated when applying the assessment model, leading to a lower TAC and a smaller impact on the stock. Given that a risk-averse utility function was applied, it therefore appeared that imperfect information led to a better management option than perfect information. See Kuikka et al. (2001) for an example where risk-neutral and business-oriented utility functions are considered.

Value-of-control

The value-of-control is related to manipulating the system in order to obtain beneficial outcomes. The expected value-of-control (EVC) can be calculated in influence diagrams by changing a node containing a probabilistic variable to a decision node, or adding a decision node in front of a probabilistic node. Part of the network can then be manipulated by this new control. EVC is the expected value of the 'new' model with new decision variable(s) less the value of the 'old' model.

EVC is a somewhat more recent concept than EVPI. EVPI has been used extensively in decision analysis modelling, but there are relatively few articles on the use of EVC in planning (Pearl, 1994). However, EVC is an essential concept in management planning. It can give insights into how much should be invested in the control of, for example, catch or fishing effort. After the control has been changed, the value-of-information can be reassessed and the monitoring system reconsidered. These kinds of repetitive steps can be essential in the planning of a system that includes both management and monitoring.

We evaluated the value-of-control on two levels: on a tactical level (simulation model) and on a strategic level (evaluations of management strategies and applying two different decision variables in the influence diagram model). On a tactical level, the value-of-control was the difference in utility of a perfect knowledge TAC run (with two alternative values: $\theta = 0.1$ or 0.3) and a "No TAC" run. This difference shows how much the additional control created by a TAC improves the situation compared to management where there are no external control mechanisms.

In the strategic model, the value-of-control was calculated by changing a random variable (like the leaving trigger) to a partly controllable variable by adding an additional decision variable to the model (see Figure 7.8). When a decision node is added before a random node, it is assumed that there is a mechanism by which the value of a previously random variable can be manipulated. The value of this additional control is the difference between the old and new versions of the model. The leaving trigger, to which the fleet reacts by stopping fishing, is assumed to be controllable by taxes or subsidies on running costs. A tax is modelled by an increase in the leaving CPUE trigger value (fishing only continues with high catch rates), whereas a subsidy is modelled by a decrease in the trigger value (fishing can continue with low catch rates). A model with no economic control was also considered.

As the utility function is risk-averse, results showed that the decision alternative "Tax" gave the highest value, and "Subsidy" the lowest. Interestingly, a combination of "Tax" and "No TAC" provides a similar outcome to "No economic control" combined with the decision rule "Perfect information, θ=0.1" (Table 7.7). This suggests that this fishery has a strong self regulatory component based on the reaction of fishers to catch rates. Economic behaviour of operators accounts for some of the management, and outcomes are not as sensitive to a successful application of the assessment model.

Table 7.7. The utility of having additional economic controls on the behaviour of the fishermen. Basic observations are assumed (first column of Table 7.6).

	Economic control applied		
Management alternative	Tax	Subsidy	No control
Fixed TAC	0.68	0.64	0.67
No TAC	0.66	0.58	0.63
Tag 500, 0.1	0.70	0.66	0.69
Perf, 0.1	0.69	0.64	0.67
Perf, 0.3	0.67	0.60	0.64

The value-of-control on a tactical level (i.e. the difference between the utility function values of "Perfect information, θ=0.1" and "No TAC") is fairly high, depending on the economic management applied. Logically, the value is highest when subsidies are applied, as the TAC is required to safeguard the SSB because there are less economic constraints. In this case, the fleet has capacity to fish the quota and the TAC is needed to keep a balance between catch and stock sustainability.

8. BENEFITS

The industry fishing for Patagonian toothfish around Macquarie Island, and Heard Island and McDonald Islands will directly benefit from this project, as will those entrusted with the management of these fisheries. Utilisation and conservation benefits will be realised through the development of appropriate harvest strategies that will facilitate the maintenance of harvested populations and marine ecosystems.

Additional benefits of the project could flow to all of the fisheries managed by AFMA as the software developed and many of the conclusions arising from the study are readily transferable to other fisheries. It should be possible to tailor the simulation framework developed as part of this project to other harvested species and regions.

9. PLANNED OUTCOMES

1. An assessment of the current status of Australian sub-Antarctic fish stocks.

The assessment of stock status of Patagonian toothfish at Macquarie Island continues to be based on the methodologies presented in this report. The assessment outputs are a critical input to the management and TAC setting process for this fishery (Tuck, 2000b; Tuck and He, 2001; Tuck et al., 2000; Tuck et al., 2001a).

2. An evaluation of various management strategies that could be adopted to facilitate management of the stock.

The results from the Management Strategy Evaluation are being used by SAFAG, industry and management to help manage the fishery in accordance with agreed sustainability objectives. The results of the project have increased fishers’ and managers’ awareness of the utility of setting and evaluating appropriate management strategies for fisheries (Tuck 2000a; Kuikka et al. 2001; Tuck et al., 2001b).

3. Improved communication and linkages between the CCAMLR and AFMA assessment process for the assessment of Heard Island and McDonald Islands Patagonian toothfish.

The application of assessment methodologies developed under this project to CCAMLR fisheries have complemented and enhanced existing CCAMLR assessment procedures. Active participation at CCAMLR and SAFAG meetings has improved communication between the AFMA and CCAMLR assessment processes (Williams et al., 2002; CCAMLR WG-FSA-00/43; CCAMLR WG-FSA-01/18; CCAMLR WG-FSA-00/49).

10. FURTHER DEVELOPMENT

10.1 Stock Assessment

The current assessment of the Macquarie Island Patagonian toothfish stock is based on the tag-recapture model initially developed by de la Mare and Williams (1997) and the extensions described here. The model uses lumped-age population and tag accumulation models to account for tags being released at various times throughout the season and the effects of removals by fishing. At this stage, length-frequency information from the catch is not directly included in the assessment, although the lengths of tagged fish are used in the assessment models that apply vulnerability functions. Incorporating length data may help refine estimates of the age-structure of vulnerable fish and provide an ability to distinguish periods of transient fish influx from those where resident fish alone are present. This will be especially important when models of 2 stocks are developed further (Appendix C). As initial results from the MSE have indicated that the spawning biomass of the resident population could be threatened by an increase in effort (by multiple operators for example) when 2 stocks are present, the development of a 2 stocks model is seen as a high priority. These kinds of models also show promise for the estimation of illegal catch, in addition to estimating available abundance of fish, from the Heard Island and McDonald Island toothfish fishery.

Applications of standardisation methodologies to catch and effort data from the trawl fishery have been applied to produce relative indices of abundance for the toothfish stocks at Aurora Trough and the Northern Valleys. The results of the standardisation showed variable trends in abundance (but consistent initial declines). Further development of the catch rate models is required to overcome some of the problems identified (such as the unbalanced nature of the dataset).

10.2 Management Strategy Evaluation

Results from an evaluation of potential management strategies for the Macquarie Island toothfish fishery appear to suggest that current fishing operations have had a smaller impact on the resource than one might expect, principally because the TAC is not caught in most years. In addition, economic constraints, which force operators to cease fishing during unfavourable catch conditions, may have the potential to conserve the resource. However, these features are reliant upon the following factors:

a) Selection is assumed to be narrow, i.e. only a fairly small proportion of the population can be affected by fishing.

b) Spawning fish, which are mostly outside of the selection range, contribute to local recruitment only (there is no emigration).

c) As fishing operations respond to a decreasing biomass through economic triggers (by having a critical mean CPUE value under which fishing is not continued), fishing has a strong self-regulatory mechanism.

d) There is only a single vessel operating over a relatively short season, i.e. effort is limited due to physical and economic constraints.

e) Estimates of virgin abundance are considered reasonable.

While these factors remain, the initial results from this work suggest that exploitation of the resource is sustainable in the long-term with minimal management intervention. However, it is highly unlikely that all of these factors are either true or will hold in the future. For example, the inclusion of longlining to the area would have a marked effect on the vulnerability function and potentially allow a greater proportion of the spawning biomass to be harvested. Likewise, if economic constraints on fishing weaken then the subsequent increased effort (also possible through multiple operators) and catch could have a substantial effect on spawning biomass.

The current management policy for toothfish is analogous to the feedback rule with $b = 0.1$ (Section 7). However, it also includes a condition that if the full TAC is taken from the resident stock, and there is no recruitment to the population, then it is possible that the resident stock has been reduced to the reference level of stock reduction (50% reduction in SSB). Under this circumstance fishing operations would cease. This 'worst-case' condition has not been included in the management strategies evaluated. However, the result showing that the resident stock biomass might reduce substantially with increased effort in a two-stocks resource scenario highlights its need. This condition should be considered in future evaluation of management strategies so that strategies are robust to all potential resource scenarios (see Appendix C).

The software developed to analyse potential management strategies is a flexible tool that, without much effort, can be utilised to consider many management options across a wide range of resource scenarios. However, there are some areas where improvements could be made. Clearly, the method used to determine catch rates is pivotal to the results. While the method used is relatively flexible, estimating three catchability-related parameters from historical data, further examination of these techniques should be considered, along with appropriate analysis of residuals. The historical component of the operating model and, in particular, the model used to condition projections should be considered in future work. This is especially the case when fitting a second stock to historical data, or determining the duration of extended selection. More detailed examination of observed length-densities may assist in this regard.

There are several more scenarios that could be examined now that the basic structure has been developed. For example, the inclusion of a spatial model would allow the analysis of management strategies when more than one stock interacts through recruitment or migration. A management strategy that includes a marine protected area could then be analysed. An appropriate model of selectivity could consider the effect of allowing longlining. As more age groups would be available to the fishery, the move from trawling to longlining may have a marked impact on the population, and hence management. Longlining may also improve estimates of growth parameters. This benefit could be considered in an evaluation of harvest strategies that include longlining. The effects of increasing the number of vessels could also be examined in more detail than was possible here. Initial analyses of increased effort showed that strong impacts could occur if the population is not managed in an appropriate manner.

11. CONCLUSION

Objective 1. To provide the SAFAG with updated information on the current status of Patagonian toothfish around Macquarie Island, including development of extended stock assessment models that link the current tag-based models to age-structured and spatially structured population models, and to commercial catch data.

Outcome:

The tag-recapture assessment models estimated that pre-tagging available biomass was between 3 and 5 thousand tonnes in Aurora Trough and between 30 and 45 thousand tonnes in the Northern Valleys region. Confidence limits on estimates of pre-tagging abundance for the Northern Valleys are very broad due to the recapture of only three tagged fish in the first season. This region shows a dramatic decline in available abundance between the first (1996/97) and second (1997/98) seasons. This reduction in available biomass is greater than can be explained by the fishery catches even in the absence of recruitment. For Aurora Trough, estimated available abundance declined over the first three seasons of tagging (to 1997/1998), but has since shown an increasing trend. This increase corresponds well to the beginning of catch restrictions for commercial fishing (a 40t research quota only was allowed) in the Aurora Trough region. For Aurora Trough, estimates of the fraction of current available abundance relative to pre-tagging levels are between 100% and 140% in numbers and 56% and 75% in terms of available biomass. Estimates of recent available abundance are most uncertain due to poor catches and the strong influence of a small number of recaptures (especially in the Northern Valleys, e.g. 1 tag recapture from 7 shots in 2000/01), and this uncertainty is directly reflected in any estimates of relative biomass.

Objective 2. To develop long-term management strategies for the Macquarie Island Patagonian toothfish fishery.

Outcome:

Results from an evaluation of potential management strategies for the Macquarie Island toothfish fishery appear to suggest that current fishing operations have had a smaller impact on the resource than one might expect, principally because the TAC is not caught in most years. In addition, economic constraints, which force operators to cease fishing during unfavourable catch conditions, may have the potential to conserve the resource.

While it is encouraging that tests of the current management strategy showed that it outperformed others, at least in terms of conservation benefits, it did falter (along with the other policies) in one key circumstance. Namely, if effort is increased markedly, there are two stocks present and no management intervention occurs when stock signals would indicate a reduction in spawning biomass. Under this scenario, fishing continues through periods of poor catch when a single operator would normally have chosen to leave. As the current stock assessment is based on models of a single population, the presence of transient fish leads to increased estimates of annual available biomass and hence TACs (to be taken from the ‘single’ population). The increased effort allows more of the TAC to be caught and subsequently more of the resident population is taken as part of the total annual catch. Through the MSE, it was shown that (with perfect information of available biomass) a TAC decision rule based on the resident stock alone could meet conservation criteria for the population. However, this initial result depended on perfect knowledge of resident available biomass, which is estimated by a stock assessment model. As such, stock assessment models of two populations, resident and transient, were developed and initial testing proved promising. A key element in these models will be the ability to distinguish between periods when available fish are a combination of residents and transients and periods of resident only fish.

A novel application of Bayesian Belief Networks is also described in this report. These methods explored the relative belief of alternative hypotheses about stock structure and movement of the toothfish population. The BBN models are based on causal networks where alternative hypotheses are linked to observable variables. In the network, all relevant information, including expert opinions, field observations, and model outputs, can be incorporated, with greater weight given to information that has less uncertainty and variance. Biological observations from genetics, tagging experiments and fatty acid compositions were included in the model. The models demonstrated how knowledge related to biological hypotheses and observations can be brought into fisheries management modelling in a relatively simple way. The models showed that the reaction of the fishers to variable daily income (through catch rates) includes a strong self-regulatory mechanism; serving to protect the stock. It was shown that, in addition to catch limits as management tools, economic tools focusing on the behaviour of the fishers can be effective in fisheries planning, especially in increasing the robustness of management systems. Moreover, changes within the utility function can reveal potential conflicts between various interest groups.

Objective 3. To participate in the stock assessments for Heard Island and McDonald Islands Patagonian toothfish and icefish conducted through CCAMLR, and assist in providing effective communication between the CCAMLR and AFMA assessment processes.

Outcome:

The 19th meeting of the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) annual Working Group on Fish Stock Assessment was held in October 2000. Dr. Geoff Tuck (CSIRO) attended as a member of the Australian Delegation. The manuscript entitled “An exact time of release and recapture stock assessment model applied to Macquarie Island Patagonian toothfish (*Dissostichus eleginoides*)” (CCAMLR WG-FSA-00/43) was presented to the working group. This paper described the assessment techniques used to assess the Macquarie Island population of Patagonian toothfish. This work was provided to promote exchanges of information on assessment methodologies at a meeting with a significant array of international experts. Dr. Tuck also provided assistance to the Incidental Mortality Arising from Longline Fishing (IMALF) working group of CCAMLR where ecological interactions involving fishing operations in the Antarctic and Sub-Antarctic are considered.

The 20th meeting of the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) annual Working Group on Fish Stock Assessment was held in October 2001. Dr. Keith Sainsbury (CSIRO) attended as a member of the Australian Delegation, participating in the assessments of icefish and toothfish. Dr. Geoff Tuck (CSIRO) also provided two background papers to the Incidental Mortality Arising from Longline Fishing (IMALF) working group, where ecological interactions involving fishing operations in the sub-Antarctic are considered. The papers by Tuck et al. “Spatio-temporal trends in longline fisheries and implications for seabird bycatch” (CCAMLR WG-FSA-01/49) and “Modelling the impact of fishery by-catches on albatross populations” (CCAMLR WG-FSA-01/18) were presented to the working group. These papers describe the effort trends in the Southern Ocean (including those in demersal and pelagic fisheries) and provides a modelling framework for assessing impacts on seabird populations.

At the September 2001 meeting of SAFAG, Dr. Tuck presented results from the first attempt at assessing the HIMI Patagonian toothfish fishery using tagging data (SAFAG 12/7). This information provided additional support to the assessments undertaken at CCAMLR. The collaborative research between CSIRO and AAD has since seen the application of the tag-recapture model to the HIMI fishery published in the journal CCAMLR Science.

Drs. Xi He and Geoff Tuck are also scientific members of SAFAG where issues related to the assessment of Australia's sub-Antarctic fish stocks are discussed by managers (AFMA), industry and scientists (CSIRO, AAD, BRS, ABARE).

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13. REFERENCES

Anon. 1994. Report of the Thirteenth Meeting of the Scientific Committee. Scientific committee for the Conservation of Antarctic Marine Living Resources. Hobart. Paragraphs 5.18 - 5.26.

Almond, R. G. 1995. Graphical Belief Modeling. Chapman & Hall. London. 427 pp.

Barndorff-Nielsen, O.E., James, I.R., Leigh, G.M. 1989. A note on a semi-parametric estimator of mortality. Biometrika 76, 803-805.

Clemen, R.T. 1996. Making hard decisions. 2nd Edition. PWS Kent, Boston. 664 pp.

Constable, A. J., de la Mare, W.K. 1996. A generalised yield model for evaluating yield and the long-term status of fish stocks under conditions of uncertainty. CCAMLR Science, 3: 31-54.

Constable, A.J., Williams, R., Lamb, T., van Wijk, E. 1999. Revision of biological and population parameters for *Dissostichus eleginoides* on the Heard Island Plateau (Division 58.5.2) based on a comprehensive survey of fishing grounds and recruitment areas in the region. Document *WG-FSA-99/68,* CCAMLR, Hobart, Australia.

Constable, A.J., Williams, R., Tuck, G.N., Lamb, T., Morrison, S. 2001. Biology and growth of toothfish. In: He, X., Furlani, D. (Eds.), Ecologically sustainable development of the fishery for Patagonian toothfish (*Dissostichus eleginoides*) around Macquarie Island: population parameters, population assessment and ecological interactions. FRDC Report 97/122, pp 87-103.

de la Mare, W.K., Williams, R. 1997. Abundance of Patagonian toothfish at Macquarie Island estimated from tagging studies. SAFAG/97/6.

Fischler, K.J. 1965. The use of catch-effort, catch sampling and tagging data to estimate a population of blue crabs. Trans. Amer. Fish. Soc. 94, 287-310.

Goldsworthy, S. D. He, X., Furlani D., Moore, T., Rintoul, S. Koslow, T., Kloser, R., Williams, D., Lewis, M., Lamb, T. 2001. Physical and biological oceanography. In: He, X., Furlani, D. (Eds.), Ecologically sustainable development of the fishery for Patagonian toothfish (*Dissostichus eleginoides*) around Macquarie Island: population parameters, population assessment and ecological interactions. FRDC Report 97/122, pp 28-58.

van der Graag, L. C. 1996. Bayesian belief networks: odds and ends. Comput. J. 39: 99-113.

Greene, W.H. 1993. *Econometric Analysis*. Macmillan, New York.

Hammond, T. R., O'Brien, C. M. 2001. An application of the Bayesian approach to stock assessment model uncertainty. ICES J. Mar. Sci.58: 648-656.

Hearn, W.S. 1986. Mathematical methods for evaluating marine fisheries. PhD thesis. University of New South Wales. 195p.

Hearn, W.S., Sandland, R.L., Hampton, J. 1987. Robust estimation of the natural mortality rate in a completed tagging experiment with variable fishing intensity. J. Cons. Int. Explor. Mer. 43, 107-117.

Hildén, M. 1997. Conflicts between fisheries and seabirds - management options using decision analysis. Marine Policy 21: 143-153.

Hoenig, J.M., Barrowman, N.J., Pollock, K.H., Brooks, E.N., Hearn, W.S., Polacheck, T. 1998. Models for tagging data that allow for incomplete mixing of newly tagged animals. Can. J. Fish. Aquat. Sci. 55, 1477-1483.

Hilborn, R., Walters, C.J. 1995. Biomass dynamic models: users manual. FAO Computerised Information Series (Fisheries) No. 10. Rome, FAO. 62p.

Jensen, F.V. 1996. An Introduction to Bayesian Networks. UCL Press. 178 pp.

Jolly, G.M. 1965. Explicit estimates from capture-recapture data with both death and immigration-stochastic models. Biometrika 52, 225-247.

Kirkwood, G.P., Constable, A.J. 2001. Integration of CPUE data into assessments using the Generalised Yield Model. CCAMLR Science, 8: 65-74.

Kuikka, S., Gislason, H., Hansson, S., Hildén, M., Sparholt, H., Varis, O. 1999. Modelling environmentally driven uncertainties in Baltic cod management by Bayesian influence diagrams. Can. J. Fish. Aquat. Sci. 56, 629-641.

Kuikka, S., Tuck, G.N., Smith, A.D.M., He, X. 2001. Using simulation results and Bayesian belief networks to assist fisheries planning: a case study of the Patagonian toothfish fishery at Macquarie Island. ICES CM 2001/N:08.

Kuikka, S., Varis, O. 1996. Uncertainties of climatic change impacts in Finnish watersheds: a Bayesian network analysis of expert knowledge. Bor. Env. Res. 2: 109 - 128.

Leigh, G.M. 1988. A comparison of estimates of natural mortality from fish tagging experiments. Biometrika 75, 347-353.

Manly, B.F.J. 2003. *Handbook on capture-recapture*. Princeton University Press. In press.

Marr, J.C. 1951. On the use of the terms “Abundance”, “Availability” and “Apparent Abundance” in fishery biology. Copeia, No. 2, pp. 163-169.

McCullagh, P., Nelder, J.A. 1989. Generalized Linear Models. Chapman and Hall, New York.

Patterson, K.R. 1999. Evaluating uncertainty in harvest control law catches using Bayesian Markov chain Monte Carlo virtual population analysis with adaptive rejection sampling and including structural uncertainty. Can. J. Aquat. Sci. 56, 208-221.

Pearl, J. 1988. Probabilistic Reasoning in Intelligent Systems: Networks of Plausible Inference. Morgan-Kaufman, San Mateo, CA. 552 pp.

Pearl, J. 1994. A probabilistic calculus of actions. In: R.L. de Mantaras and D. Poole (eds.): Proceedings of the Tenth Conference on Uncertainty in Artificial Intelligence. Pp. 454-462. San Francisco. Morgan Kaufmann.

Petersen, C.G.J. 1896. The yearly immigration of young plaice into the Limfjord from the German Sea, etc. Rep. Dan. Biol. Sta. 6, 1-48.

Polacheck, T., Klaer, N. L., Millar C., Preece, A. L. 1999. An initial evaluation of management strategies for the southern bluefin tuna fishery. ICES Journal of Marine Science, 56: 811-826.

Pope, J.G. 1972. An investigation of the accuracy of virtual population analysis using cohort analysis. ICNAF Res. Bull. 9, 65-74.

Quinn II, T.J., Deriso, R.B. 1999. *Quantitative fish dynamics*. Oxford University Press, New York.

Reilly, A., Ward, B., Williams, R. 1998. Preliminary results of investigations into the stock structure of Patagonian toothfish around Macquarie Island. SAFAG/98/4/4.

Reilly, A., Ward, B., Williams, R. 2001. Stock structure of toothfish. In: He, X., Furlani, D. (Eds.), Ecologically sustainable development of the fishery for Patagonian toothfish (*Dissostichus eleginoides*) around Macquarie Island: population parameters, population assessment and ecological interactions. FRDC Report 97/122, pp 103-108.

Ricker, W.E. 1975. Computation and interpretation of biological statistics of fish populations. Bull. Fish. Res. Bd. Can. 191. 382p.

SAFAG10, 2000. Minutes of the Sub-Antarctic Fisheries Assessment Group meeting 10, 6-7 November 2000, Hobart Tasmania.

SAFAG13, 2001. Minutes of the Sub-Antarctic Fisheries Assessment Group meeting 13, 19-20 November 2001, Hobart Tasmania.

SC-CAMLR, 1998. Report of the Seventeenth Meeting of the Scientific Committee. Scientific Committee for the Conservation of Antarctic Marine Living Resources, Hobart, Australia.

SC-CAMLR, 2001. Report of the Twentieth Meeting of the Scientific Committee. Scientific Committee for the Conservation of Antarctic Marine Living Resources, Hobart, Australia.

Seber, G.A.F. 1965. A note on the multiple recapture census. Biometrika 52, 249-259.

Smith, A.D.M., Punt A.E., Wayte, S.E., Klaer, N.L. 1996. Evaluation of harvest strategies for eastern gemfish (Rexea solandri) using Monte Carlo simulation. In, Smith A. D. M. Evaluation of harvesting strategies for Australian fisheries at different levels of risk from economic collapse. FRDC T93/238, pp 120-164.

Tuck, G. N. 2000a. Macquarie Island Patagonian Toothfish Management Strategy Evaluation Workshop: Outcomes. 3 April 2000, CSIRO, Hobart. SAFAG-00/9/. Australian Fisheries Management Authority, Canberra.

Tuck, G.N. 2000b. Standardisation of Catch and Effort Data for the Patagonian Toothfish (*Dissostichus eleginoides*) Fishery of Macquarie Island: 1995 to 2000. Report to the Sub Antarctic Fisheries Assessment Group. SAFAG-00/10/3.

Tuck, G.N., Campbell, R.A. 1999. Standardisation of Catch and Effort Data for the Patagonian Toothfish (*Dissostichus eleginoides*) Fishery of Macquarie Island: A Preliminary Analysis. Report to the Sub Antarctic Fisheries Assessment Group. SAFAG-99/7/5.

Tuck, G.N., de la Mare, W.K., Hearn, W.S., Williams R., Smith, A.D.M., He, X., Constable, A.J. 2003. An exact time of release and recapture stock assessment model with an application to Macquarie Island Patagonian toothfish (*Dissostichus eleginoides*). Fisheries Research, 63: 179-191.

Tuck, G.N., He, X. 2001. Standardisation of Catch and Effort Data for the Patagonian Toothfish (*Dissostichus eleginoides*) Fishery of Macquarie Island: 1995 to 2001. Report to the Sub Antarctic Fisheries Assessment Group. SAFAG-01/13/4.

Tuck, G.N., Williams, R. 2001. Abundance of Patagonian toothfish at Macquarie Island estimated from tagging studies: 1995-2001. Sub-Antarctic Fisheries Assessment Group document. SAFAG-01/13/5. Australian Fisheries Management Authority, Canberra, Australia.

Tuck, G.N., Williams, R., Constable, A.J., Smith A.D.M., He, X., Hearn, W.S. 1998. Abundance of Patagonian toothfish at Macquarie Island estimated from tagging studies: 1995-1998. Sub-Antarctic Fisheries Assessment Group document SAFAG/98/4/2. Australian Fisheries Management Authority, Canberra, Australia.

Tuck, G.N., Williams, R., Smith, A.D.M., He, X., Constable, A.J. 1999. Abundance of Patagonian toothfish at Macquarie Island estimated from tagging studies: 1995-1999. Sub-Antarctic Fisheries Assessment Group document SAFAG/99/7/3. Australian Fisheries Management Authority, Canberra, Australia.

Tuck, G.N., Williams R., He, X., Smith, A.D.M., Constable, A.J. 2000. Abundance of Patagonian toothfish at Macquarie Island estimated from tagging studies: 1995-2000. Sub-Antarctic Fisheries Assessment Group document SAFAG/00/10/4. Australian Fisheries Management Authority, Canberra, Australia.

Tuck, G.N., Williams, D., He, X., Smith, T., Constable, A.J., de la Mare, B., Hearn, B., Campbell, R. 2001a. Stock assessment of Macquarie Island Patagonian toothfish. In: He, X., Furlani, D. (Eds.), Ecologically sustainable development of the fishery for Patagonian toothfish (*Dissostichus eleginoides*) around Macquarie Island: population parameters, population assessment and ecological interactions. FRDC Report 97/122, pp 126-161.

Tuck, G.N., Smith, A.D.M., Constable, A., He, X., Kuikka, S., Klaer, N. 2001b. An initial evaluation of management strategies for Macquarie Island toothfish fishery. In: He, X., Furlani, D. (Eds.), Ecologically sustainable development of the fishery for Patagonian toothfish (*Dissostichus eleginoides*) around Macquarie Island: population parameters, population assessment and ecological interactions. FRDC Report 97/122, pp 162-218.

Varis, O., Kettunen, J., Sirviö, H. 1990. Bayesian influence diagram approach to complex environmental management including observational design. Comput. Stat. Data An. 9: 77-91.

Varis, O., Kuikka, S. 1997. Joint use of multiple environmental assessment models by a Bayesian meta-model: the Baltic salmon case. Ecol. Mod. 10: 341-351.

Williams, R., Lamb, T. 2001. History of the toothfish fishery. In: He, X., Furlani, D. (Eds.), Ecologically sustainable development of the fishery for Patagonian toothfish (*Dissostichus* *eleginoides*) around Macquarie Island: population parameters, population assessment and ecological interactions. FRDC Report 97/122, pp 17-27.

Williams, R., Lamb, T., Constable, A.J., Tuck, G.N. 1998. History of the Macquarie Island fishery: 1994-1998. Sub-Antarctic Fisheries Assessment Group document. SAFAG-98/4/6. Australian Fisheries Management Authority, Canberra.

Williams, R., Tuck, G.N., Constable, A.J., Lamb, T. 2002. Movement, growth and available abundance to the fishery of *Dissostichus eleginoides* Smitt, 1898 at Heard Island, derived from tagging experiments. CCAMLR Science, 9:33-48.

Wilson, G., Nichols, P. 2001. Fatty acid analysis of toothfish. In: He, X., Furlani, D. (Eds.), Ecologically sustainable development of the fishery for Patagonian toothfish (*Dissostichus eleginoides*) around Macquarie Island: population parameters, population assessment and ecological interactions. FRDC Report 97/122, pp 241-274.

Xiao, Y. 1996. A general model for estimating tag-specific shedding rates and tag interactions from exact or pooled times at liberty for a double tagging experiment. Canadian Journal of Fisheries and Aquatic Sciences, 53: 1852-1861.

Xiao, Y., McShane, P. 2000. Estimation of instantaneous rates of fishing and natural mortalities from mark-recapture data on the western king prawn *Penaeus latisulcatus* in the Gulf St. Vincent, Australia, by conditional likelihood. Trans. Am. Fish. Soc. 129, 1005-1017.

APPENDIX A – INTELLECTUAL PROPERTY

No intellectual property has arisen from the project that is likely to lead to significant commercial benefits, patents or licenses.

APPENDIX B – STAFF

Geoff Tuck	Resource Modeller	50%
Xi He	Senior Research Scientist	10%

APPENDIX C – TWO-STOCKS MODELS

Geoff Tuck and Keith Sainsbury

The dramatic decreases in apparent abundance of Patagonian toothfish observed at Macquarie Island has led to the hypothesis that a second ‘transient’ stock occasionally frequents the region. Its appearance leads to substantial increases in catch and catch rates. The 1996/97 season in the Northern Valleys is especially noteworthy for its exceptional catches and equally noteworthy for the apparent disappearance of the stock in subsequent years. The estimated decline in abundance is substantially greater than can be explained by the removals from fishing.

Past assessments have assumed a single reproductive stock existed in the area and that declines in available abundance were due to local movement of the stock, perhaps to deeper or untrawlable ground. However, following the results of the Management Strategy Evaluation (MSE) presented in Tuck et al. (2001b) and section 7.1, where various harvest strategies were tested against potential ecological scenarios, it became clear that an assessment was required that considered the possibility of a second stock. Evaluation of the current feedback management strategy indicated that, while it was able to conserve the stock in most cases, if a transient population existed and effort increased substantially above that currently employed, the ‘resident’ stock could be severely impacted (Figures 7.2 and 7.3). This was because the second stock maintained catch and catch rates and thus enabled the vessel (or vessels) to remain on the grounds - all the while gradually eroding the resident stock biomass.

The models presented here are a first attempt at assessing the status of a resident stock, assuming a transient stock exists. In time this assessment model will be considered as part of a management strategy for testing within the MSE framework developed (Tuck et al., 2001b). Appropriate decision rules, which utilise the results from the ‘two-stocks’ assessment, can then be tested against various ecological scenarios, including the two stocks model (but importantly also the single stock model so that an analysis can be made of the impact on management objectives of applying the wrong ecological hypothesis in the stock assessment).

Back projecting with zero recruitment

This simple model applied to the first season in the Northern Valleys assumes that there has been no recruitment between the first and second seasons and that all of the fish in the second season are resident fish. The available biomass is estimated by projecting the available abundance at the beginning of the modelled second season (1 July 1997) backwards, adding on natural mortality and catch. A similar method to estimate pre-tagging available abundance of the resident stock can be found by rearranging the following equation for B_0,

$$B_1 = (B_0 e^{-M/2} - C)e^{-M/2} \tag{C.1}$$

$$B_0 = (B_1 e^{M/2} + C)e^{M/2} \tag{C.2}$$

where B_1 is the available biomass at 1 July 1997, is the pre-tagging available biomass and C is the catch of resident fish in the first season. Assigning the catch to resident or transient fish is problematic. Assume that either all of the fish caught (C = 500t) are resident or all are transient. To illustrate the model we have used assessment model LDS with M = 0.1 in order to estimate the available biomass at the beginning of the second season (B_1 = 2430t). With resident catch zero, equation (C.1) gives B_0 = 2680t, and with all catch resident B_0 = 3211t. Figure C.1 shows the back projected available biomass trajectories. This figure also highlights the large drop in available biomass between seasons and the relatively low magnitude of catch in comparison. The model is clearly inadequate in a number of key areas, namely: it cannot estimate recruitment, nor differentiate between resident and transient catch and makes no use of tagging information in the first season.

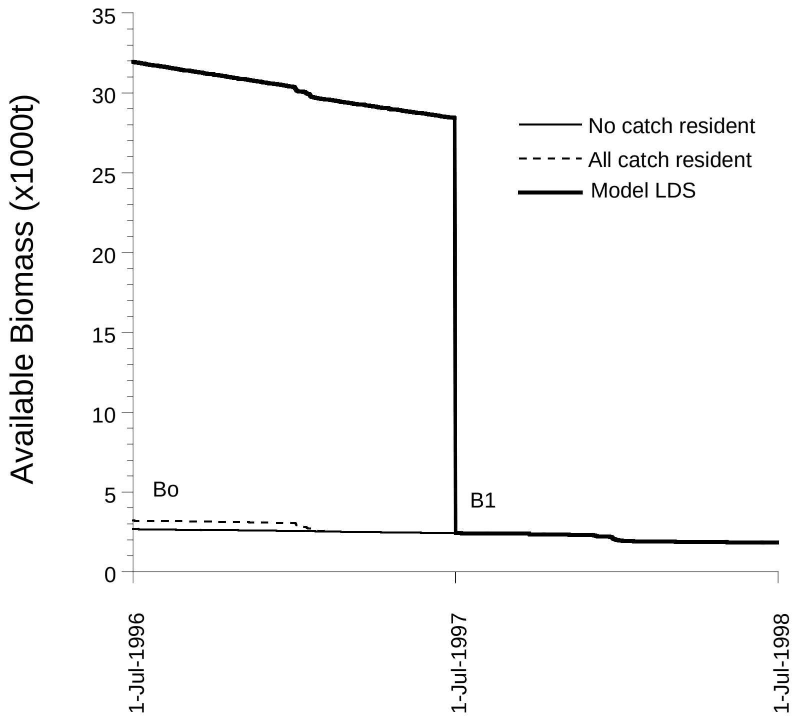


Figure C.1. The trajectory of available biomass for the Northern Valleys using model LDS and two back-projecting methods. B_0 is the estimated pre-tagging available biomass, and B_1 is the available biomass at the beginning of the modelled second season (1 July 1997). The dashed line assumes that all catch in the first season was from a resident population, whereas the thin solid line assumes no catch from resident fish was taken.

Tag-recapture assessment on resident fish alone

This model assumes that there are periods in a season when only resident fish are being caught and tagged. These may or may not be followed by periods where resident and transient fish are mixed. The basic assumption is that the tag-recapture model can estimate the available abundance of the resident stock from the resident-tagged fish only. If there is mixing of the stocks then it becomes difficult to separate tagged fish from each population. Figure C.2 shows a hypothetical example where it may be possible to estimate the available abundance of the resident stock. The available abundance can be estimated from time t_0 to t_1 as it is assumed the fished population, and all tagging and catch, is resident only over this period. Similarly available abundance from time t_2 to t_3 can be estimated by removing all catch and tagging information between times time t_1 to t_2 during which transient fish are present. The tagging data from the initial period can be included in the estimation. The model is then run from time t_2 to t_3.

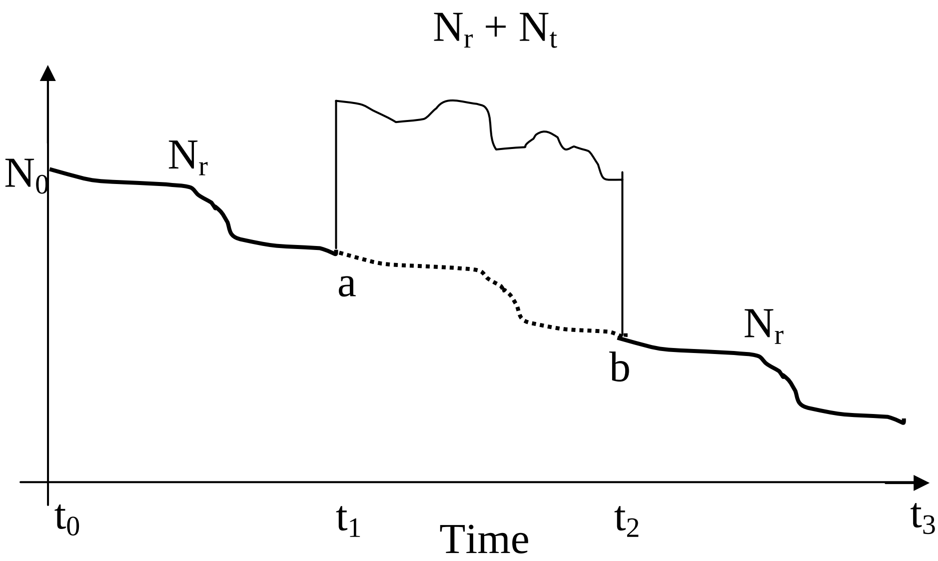


Figure C.2. A hypothetical trajectory of available numbers of resident N_r and transient N_t fish during a single season.

There are clearly problems with this method. First, a satisfactory rule needs to be developed that assigns periods to resident only fish and mixed fish. If the rule allows frequent interchange between resident and mixed stocks, then there may be insufficient tagging data on resident only days. For illustrative purposes, data from catch length-frequencies were used as a means to assign the presence or absence of transient fish on a particular day or period of days. Figures C.3 and C.4 show for each ground the daily proportion of fish greater than 700mm in the catch and corresponding catch rates. It may be possible to assign a rule to the presence or absence of transient fish according to the size of the fish in the catch. For example, if the proportion of fish in the catch greater than 700mm on a given day (or days) is greater than 50% then assume that there are transient fish in the water. Figures C.3 and C.4 show that a simple rule such as this may work in some instances but not particularly well at other times.

This model also relies upon there being a sequence of resident-mixed-resident stocks in the water. Any other sequence and estimation of the resident stock cannot be achieved for the full season. Back or forward projecting may be used for instances where the sequence is resident-mixed or mixed-resident, however assigning catch to one or the other populations in this case is problematic.

The estimation of resident abundance with this model also requires releases and recaptures to occur during periods of resident only fish. In the Northern Valleys' first season, there were only 3 tagged fish recaptured. These three fish were recaptured in late January 1997 – a period most likely to be assigned as 'mixed' due to high catch rates and a large proportion of big fish (Figure C.3, period B). As no fish were recaptured during resident only periods (A and C), the resident stock cannot be estimated with this method during the first season. Similar problems occur if we wished to apply the model to Aurora Trough (although it is generally considered that the transient population model is not applicable to this region). There were 2 recaptures in period G, 6 in H and 34 more until the end of the season (Figure C.4). Only 2 recaptures during the period G (and none prior to 3 January) leads to the usual problem of imprecision in parameter estimates (Figure C.5).

Figure C.5 illustrates an example of this model applied to the first season in Aurora Trough using periods G, H and I as defined above. The recapture of only 2 tagged fish in the first period has led to a large estimate of available abundance from time t_0 to t_1 (period G). This is then countered by a small estimated abundance from time t_2 to t_3 (period I). This example produces a decline of 3,400t between t_1 and t_2 (excluding natural mortality). Less than 1000t in total were taken from the Aurora Trough region in 1995/96. The level of decline cannot be accounted for through natural mortality and catch, and is unlikely to be due to a change in availability over the 30-day period.

A two-stocks tag-recapture model

This model attempts to use all of the tagging information by assigning a proportion of the available fish to transient and resident populations on each day of fishing.

Assume that on day *t* there are a total of N_t^T fish available, composed of resident fish N_t^r, and transient fish N_t^s,

$$N_t^T = N_t^r + N_t^s. \tag{C.3}$$

Let λ_t^s be the fraction of the available fish in the water on day *t* that are from the transient population,

$$\lambda_t^s N_t^T = N_t^s, \tag{C.4}$$

and therefore,

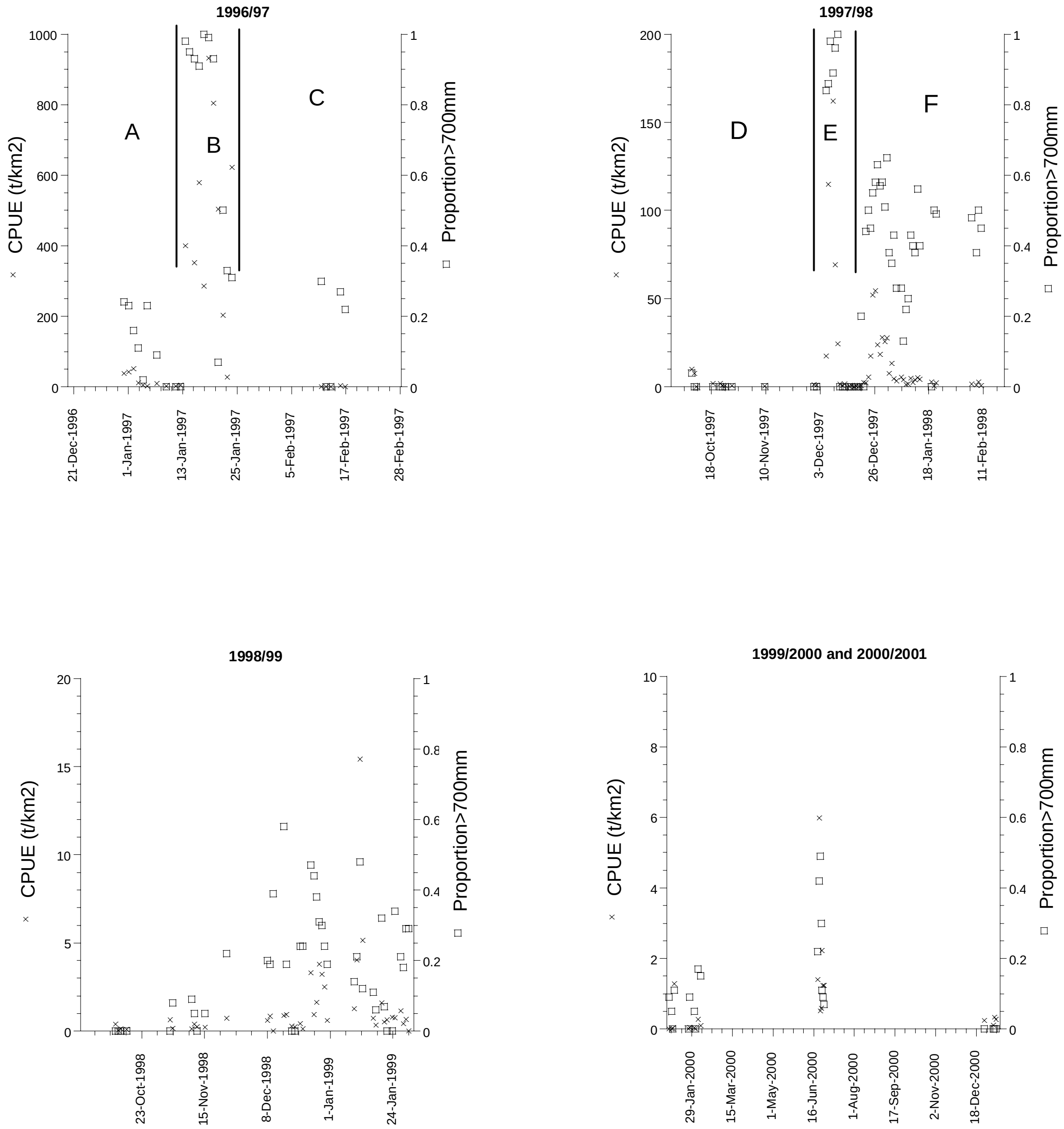
$$N_t^T = \frac{N_t^r}{1-\lambda_t^s}. \tag{C.5}$$

Let the number of resident fish available on day *t* be given by the recursive population model,

$$\begin{aligned} N_t^r &= (N_{t-1}^r - C_{t-1}^r)S \\ &= (N_{t-1}^r - (1-\lambda_{t-1}^s)C_{t-1}^T)S \end{aligned} \tag{C.6}$$

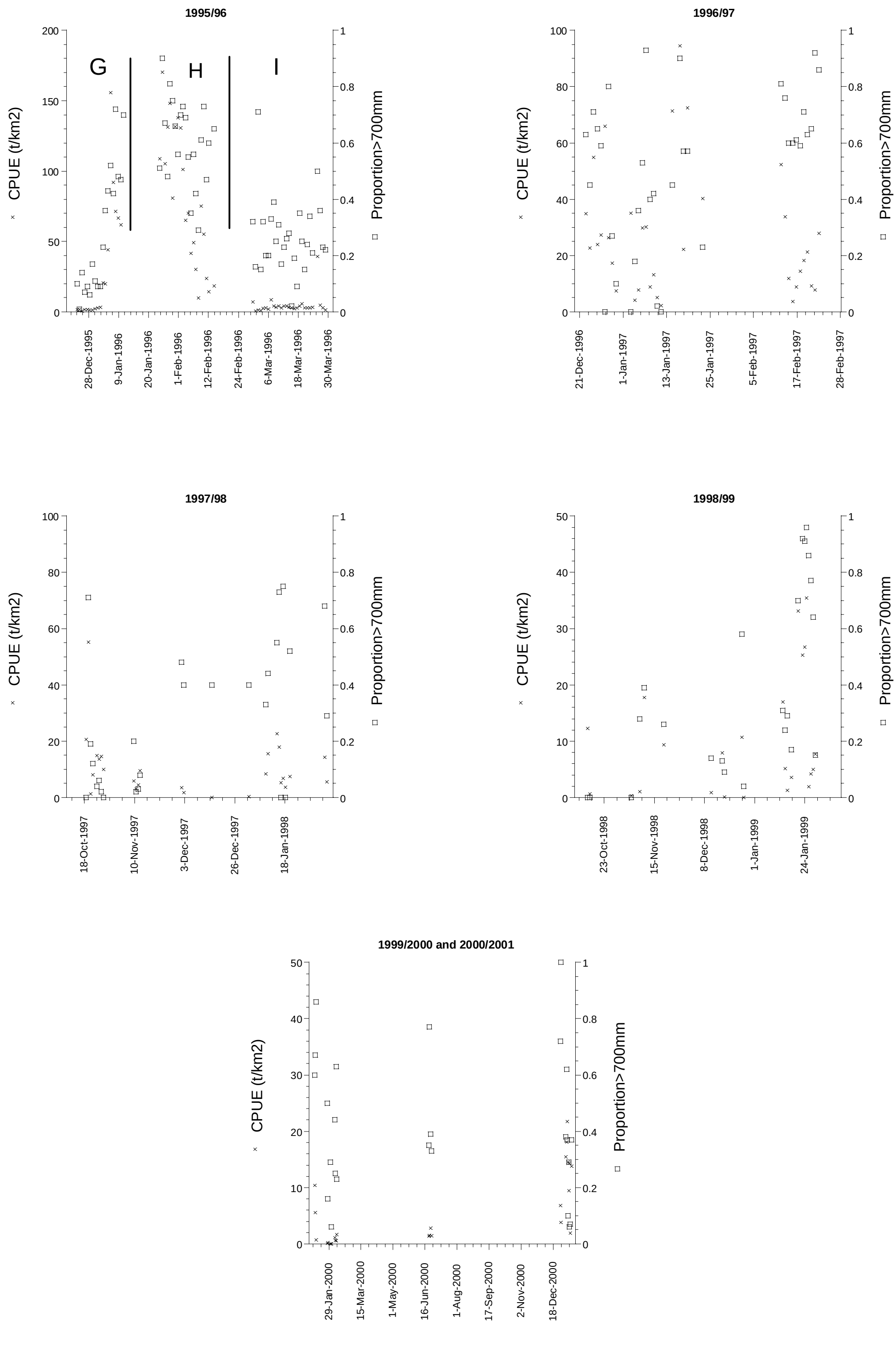
where C_t^r and C_t^T are the catch from the resident population and combined populations respectively, and $S = exp(-M/365)$ is the natural survival rate. Net recruitment to the resident population R_y occurs on 1 July.

Assuming no bias in the tagging proportions between resident and transient fish, the expected number of tagged fish released from the resident and transient populations are $(1-\lambda_t^s)p_t^T$ and $\lambda_t^s p_t^T$ respectively, where p_t^T are the total number of tagged fish that are released on day t.



Period	From	To	Releases	Recaptures
A	1/1/97	13/1/97	201	0
B	14/1/97	24/1/97	293	3
C	25/1/97	17/2/97	46	0
D	10/10/97	5/12/97	266	21
E	6/12/97	11/12/97	251	13
F	12/12/97	10/2/98	20	95

Figure C.3. For the Northern Valleys, shown are the daily proportion of fish in the catch (including tagged fish) that are greater than 700mm in length and the corresponding catch rate.



Period	From	To	Releases	Recaptures
G	24/12/95	16/1/96	84	2
H	17/1/96	16/2/96	156	6
I	17/2/96	29/3/96	250	34

Figure C.4. For the Aurora Trough, shown are the daily proportion of fish in the catch (including tagged fish) that are greater than 700mm in length and the corresponding catch rate.

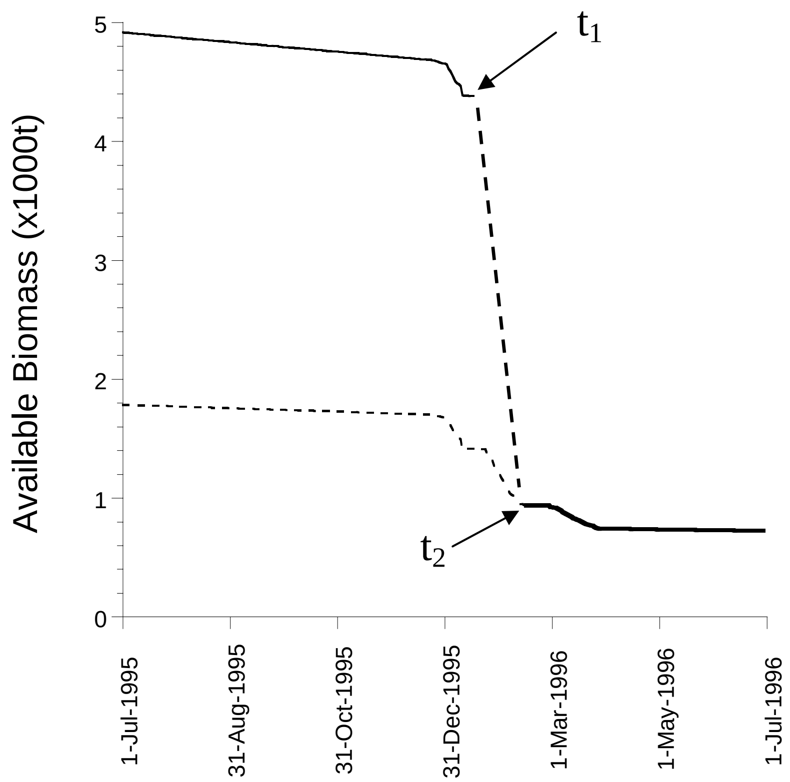


Figure C.5. Estimated trajectories of available abundance for a resident stock in the first season in Aurora Trough. The estimated decline in available abundance is greater than the total catch from the fishery during this season.

Following from equation (5.8), the expected number of surviving available tagged fish on day *t*, m_t^T, is the sum of the expected number from both resident and transient populations. The expected number of tagged resident fish in the water is the sum over all tagged fish *j* of the probability that the tagged fish was from the resident population when released, multiplied by the probability that it is still alive on day *t*. This is then multiplied by the probability that it is available on day *t*, i.e. the vulnerability. A similar equation exists for transient fish, giving

$$m_t^T = \sum_j s(l_t^j)(1-\lambda^s_{t^j_{rel}})\exp(-(t-t^j_{rel})M/365) - \sum_i \frac{1}{\omega_{t^i_{rec}}} s(l_t^i)(1-\lambda^s_{t^i_{rel}})\exp(-(t-t^i_{rec})M/365)$$
$$+ I_{\lambda_t>0}\left(\sum_j s(l_t^j)\lambda^s_{t^j_{rel}}\exp(-(t-t^j_{rel})M/365) - \sum_i \frac{1}{\omega_{t^i_{rec}}} s(l_t^i)\lambda^s_{t^i_{rel}}\exp(-(t-t^i_{rec})M/365)\right)$$

(C.7)

where the summations over index j relate to all releases (recaptured or not) prior to day t and the summations over index i relate to only recaptured fish prior to day t. The terms t_{rec}^{j} and t_{rel}^{j} are the day of recapture and release respectively of tagged fish j and $\lambda_{t_{rel}^{j}}^{s}$ is the proportion of transient fish that were available on day t_{rel}^{j}. The term $I_{\lambda_t > 0}$ is an indicator function taking the value 1 if transient fish are assumed to be present on day t and 0 otherwise. The bracketed expression following the indicator function gives the expected number of transient marked fish when transient fish are present.

From equations (C.5) and (5.5), the expected number of recaptures of both resident and transient fish becomes,

$$\mu_t = E[r_t^T] = \omega_i \frac{m_t^T}{N_t^r} C_t^T (1 - \lambda_t^s), \tag{C.8}$$

where T refers to the combination of resident and transient fish.

As there are large catches (or samples), the Poisson distribution approximates the binomial distribution, and thus $r_t \sim Po(\mu_t)$, inferring random, non-clumped returns.

Maximum likelihood estimates of the pre-tagging available abundance of the resident population, N_0^r, and net recruitment in year y, R_y^r, are then found by maximising the log-likelihood function given by

$$\ln L(r; N_0^r, R_y^r) = \sum_{t:\mu_t \neq 0} (r_t \ln(\mu_t) - \mu_t). \tag{C.9}$$

An appropriate means of determining when transient fish are present is needed for this model to be applied. More detailed analyses of catch rates and length frequencies may prove fruitful in discriminating the stocks.

APPENDIX D – BENEFICIARY RESPONSES

AUSTRAL FISHERIES PTY. LTD.

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Dr G Tuck
CSIRO Marine Laboratories
Castray Esplanade
Hobart
TAS 7000

November 2003

Geoff

Dear Dr Tuck

I am writing on behalf of Austral Fisheries, in relation to the FRDC study that you undertook , called "Stock assessment and Management Strategy Evaluation for sub-Antarctic Fisheries" (FRDC/ARF project 200/109).

I wanted to confirm Austral Fisheries' strong support for your research and its outcomes, and to thank you for your efforts in this project. Specifically, I wanted to note our appreciation for your professionalism, as well as your willingness to share the outcomes and explain them to industry, scientists and managers alike. I believe the work has been invaluable - in particular for the Macquarie Island toothfish fishery, and its' future management.

The work has proven to be the fundamental basis for management of the fishery currently, and has demonstrated a means of moving forwards with stock assessment and management of the toothfish fisheries generally. I am sure it will have application to other fisheries in future, and will continue to work with yourself and others involved in stock assessments for sub Antarctic fisheries.

Best of luck with future programs.

Kind regards

Martin Exel
General Manager, Environment and Policy
Austral Fisheries Pty Ltd

SouthMAC

Sub-Antarctic Fisheries Management Advisory Committee

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Canberra Business Centre
ACT 2610

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9 December 2003

Mr Crispian Ashby
Project Manager – Research
Fisheries Research and Development Corporation
PO Box 222
WEST DEAKIN ACT 2600

Dear Mr Ashby

RE: Stock Assessment and Management Strategy Evaluation for Sub-Antarctic Fisheries

I am writing on behalf of the Sub-Antarctic Fisheries Management Advisory Committee (SouthMAC) and the Sub-Antarctic Fisheries Assessment Group (SAFAG) in relation to the above FRDC Project, 2000/109.

This report and the previous reports prepared by Dr Geoff Tuck et al have been very useful to the management of the Macquarie Island Fishery, and the assessment is used to develop the annual Patagonian toothfish total allowable catches for the Aurora Trough and Northern Valleys regions of the Fishery.

Importantly the work will be used in the development of a formal Management Plan for the Macquarie Island Fishery, and the Strategic Assessment Report that must be prepared for the Fishery. Much of the work undertaken in the previous project 'Ecologically sustainable development of the fishery for Patagonian toothfish (*Dissostichus eleginoides*) around Macquarie Island' has direct application to address the Terms of Reference for the Strategic Assessment Report, and the results from Project 2000/109 will further enhance our knowledge on the Fishery.

The Macquarie Island Trawl Fishery has been subject to variable stock projections in recent years, and the model has been very beneficial in predicting potential yields in future years to meet decision rules developed by SAFAG.

Both SouthMAC and SAFAG commend the Report to FRDC.

Yours sincerely

W. Nagle

Bill Nagle
Chair
SouthMAC