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Preliminary investigations into the feasibility  
of assessment models for  
New Zealand longfin eels (*Anguilla dieffenbachii*)

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## EXECUTIVE SUMMARY

**Dunn, A.; Beentjes, M.P.; Graynoth, E. (2009). Preliminary investigations into the feasibility of assessment models for New Zealand longfin eels (*Anguilla dieffenbachii*). *New Zealand Fisheries Assessment Report 2009/30*. 42 p.**

The status of the New Zealand longfin eel (*Anguilla dieffenbachii*) stock is not known, but is believed to be in decline. The most recent New Zealand Plenary Report reported that “Estimates of current and reference biomass [for New Zealand longfin eels] are not available”, but also notes that “there is a high risk that the current exploitation levels for longfin eels in particular, coupled with past and present anthropogenic impacts, are not sustainable”.

Development of integrated stock assessment models can provide a basis for developing programmes of systematic data collection and their subsequent interpretation, as well as a basis for recommendations of catch levels. This report investigates the feasibility of using conventional, but suitably structured, population dynamics models that may provide a basis for discussion of programmes of systematic data collection and their subsequent interpretation.

While we report the results of the first attempt at developing an age structured population model for the longfin eel fishery, we note that the model estimates presented here investigate only a limited range of model assumptions and hypotheses that may be plausible for longfin eel populations. In particular, the choice of modelling Southland data alone is not ideal, particularly if the spawning stock biomass of longfin eels is made up from Fishstocks throughout New Zealand. Two spatial models structures were investigated— single-area and two-area. The single-area models ignored that part of the population believed to reside in closed areas or areas that otherwise had lifelong protection from fishing. The two-area models assume a constant proportion of recruitment to a closed area. We note that the two-area approach can easily be extended to encompass a larger number of areas, including Fishstocks and areas with different exploitation histories.

Estimates from the models in this report suggested that estimated pre-exploitation spawning stock biomass ranged between about 120 and 180 t, with the biomass of legal sized eels between 1700 t and 2100 t. Current spawning stock biomass was estimated to be either about 5% or 20–25% , depending on whether a single-area or a two-area model was used. Simple projection suggested that, for the single stock models and at current catches, the SSB estimates from most models stayed at similar levels or declined slightly. The exception to this was when we assumed no stock-recruitment relationship — here, the stock underwent some rebuilding.

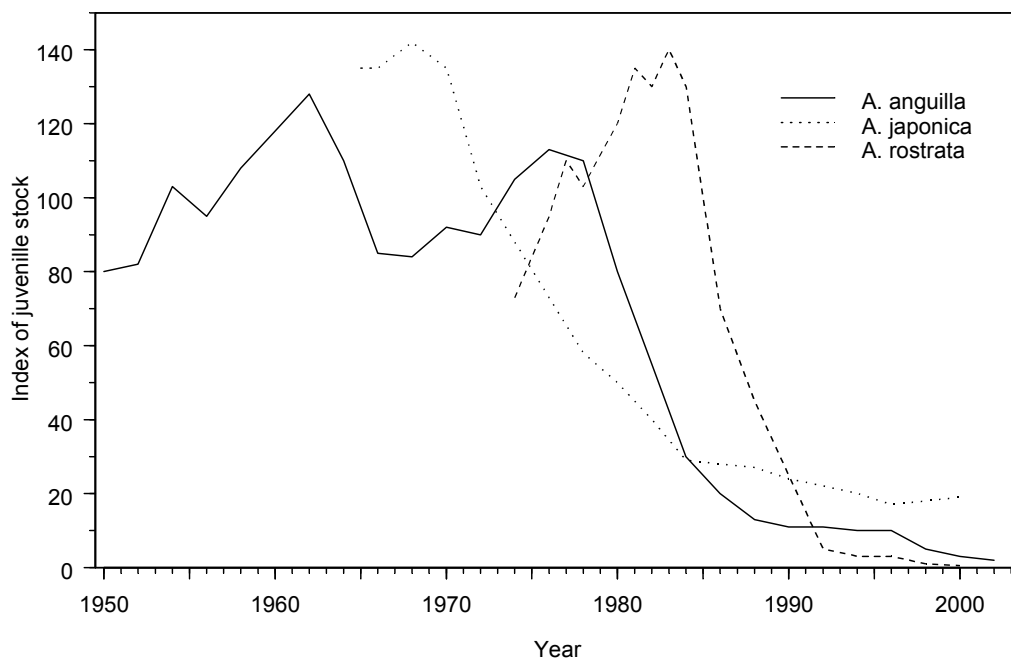
In general, we conclude that suitably structured models should be able to provide useful summaries of the current state of longfin eel populations, if the biological assumptions within the model are broadly correct, the data or observations reflect underlying population processes, the simplifying assumptions are valid, and if the models can be modified to recognise different areas with different exploitation histories. However, we also note that additional data collection and analysis to investigate these assumptions will be needed to resolve some of these issues.

# 1. INTRODUCTION

## 1.1 Background

The status of the New Zealand longfin eel (*Anguilla dieffenbachii*) stock is not known, but is believed to be in decline. The most recent New Zealand Plenary Report (Ministry of Fisheries Science Group 2006) reported that “Estimates of current and reference biomass [for New Zealand longfin eels] are not available”, but also notes that ‘there is a high risk that the current exploitation levels for longfin eels in particular, coupled with past and present anthropogenic impacts, are not sustainable’.

In general, eel stocks world wide are believed to be in decline and there are major concerns for the sustainability of fisheries for the American eel (*Anguilla rostrata*), the European eel (*A. anguilla*), and the Japanese eel (*A. japonica*) (Figure 1). In some cases, recent recruitment has declined by as much as 99% since 1980 and adult populations have also fallen dramatically (Allen et al. 2006, Casselman 2003, Clarke 2003, Dekker 2003a, 2003b, International Council for the Exploration of the Sea 2005, Richkus & Whalen 2000, Stone 2003, Tatsukawa 2003). As growth is slow for most eel species, eels are long-lived, and spawning occurs only once, it has taken some time for those fisheries to have shown signs of over fishing and recruitment decline. The decline in recruitment of *A. anguilla* was preceded by a decline in landings by about 20 years, with some authors suggesting that there may now be insufficient spawning stock to maintain the population (Dekker 2003a). The 2005 report of the ICES/EIFAC working group on eels (WGEEL) (International Council for the Exploration of the Sea 2005) concluded that the available data base for analytical assessment of the [eel] stock[s] and impacts on it were inadequate, but recommended further development of methodologies to assess stock status.



**Figure 1: Trends in juvenile abundance of the European (*A. anguilla*), American (*A. rostrata*), and Asian eel (*A. japonica*). The index is the percent of stocks in the 1960s and 1970s. (Reproduced from figure 1 in Anonymous 2003.)**

The complex life cycle of eels, involving a marine spawning migration followed by a passive return of leptocephali larvae to freshwater, make it difficult to determine what has been causing the declines in eel species worldwide. In freshwater, factors that may have impacted eel populations include pollution, fishing pressure, loss of habitat, competition with introduced species, and obstructions to migration. In the marine phase of the life cycle, changes in climate may be influencing the return of larvae to freshwater environments of the country of origin. For example, there is some evidence that

the Gulf Stream has weakened and that this may have been responsible for the decline of both the American and European eels that rely on the current to transport the larvae from the spawning grounds in the Sargasso Sea, south of Bermuda, to the eastern seaboard of North America and to Europe, respectively (Castonguay et al. 1994). Within New Zealand, the impact of currents on migration and recruitment is not known however Jellyman et al (1996) suggested that changes in oceanic currents may explain the increase in reports of the Australian longfin eel (*A. reinhardtii*) in the north of the North Island.

In New Zealand, although there has been some optimism that eel populations are less severely affected than those from Europe, Asia, and North America, there are nonetheless concerns that the longfin eel fishery may have led to a decline in spawning stock abundance, and hence eel recruitment (Hoyle & Jellyman 2002, Jellyman et al. 2000). Trends in catch-per-unit-effort (CPUE) throughout New Zealand (Beentjes & Bull 2002, Beentjes & Dunn 2003a) and observations of changes in the length and sex ratios of eel populations from the main river fisheries suggest that New Zealand's eel stocks have declined significantly over the past 15 years. While Burnet (1952) found that between 45% to 90% of longfin eels in the main Southland rivers were large females in the early 1950s, recent studies suggest that most of the fishery-recruited longfin eels are now predominantly male (Beentjes 1999, 2005, Beentjes & Chisnall 1997, 1998, Beentjes et al. 2006, McCleave & Jellyman 2004).

However, unlike for Europe and North American eel stocks, there are no long-term recruitment indices available for New Zealand eels. Programmes that have monitored both annual glass eel recruitment (Jellyman et al. 2002, Jellyman & Sykes 2004) and elver migrations (Martin et al. 2005, Martin et al. 2004) are relatively short, and were implemented in the mid 1990s. While these time series do not suggest that the level of longfin recruitment has declined over the last 10 years, anecdotal reports from fishers suggest that recent observed elver runs are significantly less dense and frequent than they once were.

In general, systematic collection of information on population size and structure, productivity, exploitation rates, natural mortality, recruitment, and spawning escapement allows development of scientific methods that can inform management of a fishery. For New Zealand eels, observed trends in CPUE indices, influenced by recorded changes in the levels of commercial catches, observations of the proportions of eels within different commercial weight grades, catch-sampling of commercial landings, and ad-hoc biological sampling of selected rivers have provided the basis for management advice. However, the New Zealand Plenary Report (Ministry of Fisheries Science Group 2006) noted that "...there are no stock assessments, or reliable data or time series, on which to base specific recommendations on catch levels".

Development of integrated stock assessment models can provide a basis for developing programmes of systematic data collection and its subsequent interpretation, as well as a basis for recommendations of catch levels. The Ministry of Fisheries Plenary Report (Ministry of Fisheries Science Group 2006) reported that conventional stock assessment methods cannot be applied to eels (noting that "standard population models are inadequate to describe the unusual life cycle of freshwater eels"), and there have been few attempts to develop modern integrated population dynamics models for New Zealand eel stocks.

This report investigates the feasibility of using conventional, but suitably structured, population dynamics models that may provide a basis for discussion of programmes of systematic data collection and their subsequent interpretation. We provide a synthesis of currently available abundance and fishery-related data and discuss how these may be included within an age-structured population model for Southland longfin eels. Further, we use these data within a preliminary model to assess the potential problems with such an approach. However, we caution the reader that results presented here are predominantly for illustrative purposes, and may not be an accurate reflection of the current state of the Southland longfin eel population — some of the data used within these models show conflicting trends that have yet to be resolved, and further, these models are based on parameters and simplifying assumptions that have yet to be tested or validated.

This report is in partial fulfilment of Project EEL2004/04, Objectives 1 “to collate and summarise the available data on stock structure, recruitment, maturity, mortality, catch, length and age composition, and abundance of South Island longfin eels suitable for inclusion into an age-based population dynamics model”, and 2 “To assess the feasibility of developing a population dynamics model for South Island longfin eels.”

## **1.2 Review of modelling studies on eels**

### **1.2.1 New Zealand studies**

The first attempt at modelling the New Zealand eel fishery was a simulation study by Francis & Jellyman (1999) that investigated if mean length data alone could be used to detect a change in biomass in eel populations, and thus estimate fishing pressure. They used data and biological parameters from the 1996<sup>1</sup> and 1997 catch sampling programmes, NIWA freshwater fish database, and other published information on New Zealand eels and northern hemisphere eel species. They concluded that length data alone were inadequate for monitoring short-term changes in eel populations, but may be useful for detecting large changes to stock biomass over the long term. They also noted that the addition of age data would be worth exploring as an additional input in future studies.

Building on the model of Francis & Jellyman (1999) and using essentially the same data, Hoyle & Jellyman (2002) examined the effects of commercial harvest on escapement of migrating New Zealand eels. They used simulations to estimate how fishing may affect spawning per recruit and yield per recruit, and to investigate the effects of alternative management actions (for example, alternative minimum and maximum size limits). They reported that with exploitation rates of 5% and 10%, at the current minimum legal weight of 220 g, the spawning biomass would be reduced by 83% and 96.5%, respectively from the initial, equilibrium state. They concluded that, at the then current level of exploitation and minimum legal weight, longfin female eels were severely recruitment overfished.

Jellyman et al. (2000) used the same basic model as Francis & Jellyman (1999), but incorporated a wider and more comprehensive range of data from relatively recent biological sampling and from historic data as early as 1948. They reported that the cropping rate [exploitation rate] might be as high as 20% in fished areas and, that at that level, few females would survive to spawn. They concluded that longfin eels were overfished and that this had resulted in a significant reduction in recruitment.

Graynoth & Niven (2004) employed Geographic Information System (GIS) methods to estimate biomass of eels in New Zealand catchments. They developed a regression model, based on individual river mean annual flows and gradients using field data on the local biomass of eels collected from a wide variety of rivers and streams in the west coast South Island and Southland. They reported that the regression explained 64% of the variation in eel biomass over the sampled sites ( $n = 130$ ). They concluded that the approach was ‘a superior measure of habitat to suitability-of-use curves and other indices’. Using their model, Graynoth & Niven (2004) provided estimates of the biomass of longfin eels, inside and outside protected areas, on the West coast South Island and in Southland.

Doole (2005) applied a bio-economic model, developed for the clam fishery in the United States (Conrad 1982), to the longfin eel fishery in the Waikato River. The main objective of the modelling was to investigate the optimal management policy for the longfin eel fishery. Doole’s (2005) model considered the economic relationships of the dynamic effects of density dependent growth and spawner-recruitment relationships. The conclusions of status were similar to those of Hoyle &

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<sup>1</sup> Note that this report uses the Ministry of Fisheries fishing year (1 October to 30 September), and we use the term ‘year’ as a label by the later calendar year in the year range, i.e., we use the label 2006 to refer to the fishing year 2005–06.

Jellyman (2002), that harvest levels based on historical catches were probably unsustainable and that area closures were required to allow sufficient spawning escapement of female longfin eels.

### 1.2.2 International studies

There are few stock assessment models for eels published in the international literature, possibly reflecting difficulties in producing useful assessments for species for which there are generally few data on spawning stock biomass, spawning escapement, and recruitment. And for European eels, may also reflect difficulties in undertaking traditional Virtual Population Analyses (VPA) for where annual catch-at-age data are not typically available. This is further complicated by the diadromous life cycle of eels which spawn only once, highly variable and local density dependent growth and natural mortality, and poorly understood factors determining gender differentiation. The studies described below all relate to the European eel, *A. anguilla*.

Vøllestad & Jonsson (1988) investigated population dynamics, productivity, and growth of *A. anguilla* from the Imsa river in Norway between 1975 and 1987. Recruitment of elvers and migration of silver eels were recorded along with annual catch over the 13 years. Length and age were also collected and these data were used to estimate total mortality, to test for density dependent growth mechanisms, and to model predicted yields of migrating eels. They showed that survival rate was density dependent, that mortality was 73% over 13 years ( $Z=0.167$ ), that there was a significant relationship between numbers of elvers ascending the river and migrating eels 8 years later, and the sex ratio was skewed strongly in favour of females. They reported that their results using a yield model agreed well with yield estimated from the recruitment escapement data over the same time period.

De Leo & Gatto (1995) used a demographic age and length structured model which they applied to the *A. anguilla* fishery of the valli di Comacchio lagoons in Northern Italy. They estimated mortality, rate of metamorphosis of yellow to silver eels (migrating), and abundance and biomass for yellow and silver eels, by sex, length, and age. Their model allowed parameters such as survivorship, net selectivity, and maturity to be modified to test alternative hypotheses on the fisheries dynamics, as well as capable of being used to investigate fisheries management policies and the effects of restocking. Using this model, De Leo & Gatto (1996) estimated recruitment to these lagoons along with density of immature eels. As density in the lagoons had declined (through fishing), the model indicated an increased survival rate as well as an increase in length and age at maturity. The results were attributed to the effects of density dependence and the trade-off between maximising reproductive potential and survival. The authors later carried out a bio-economic analysis of the silver eel fishery in the Comacchio lagoons using a demographic model including age, size, and environmental stochasticity (De Leo & Gatto 2001). The results indicated that most economic benefit is obtained by continuing to fish silver rather than yellow eels, and by restocking with elvers, but not to the point where density-dependent mortality outweighs the benefits of restocking.

In the absence of a unified European eel stock assessment, and given the pessimistic outlook for the *A. anguilla* fishery throughout continental Europe, a cohort model based on life-stage rather than length or age was attempted using catch data and other estimates such as proportions of yellow and silver eels, age and size at migration, escapement of silver eels, natural mortality, restocking data etc., (Dekker 2000b). The model was based on a modified Beverton-Holt population model (Beverton & Holt 1957) and estimated fishing mortality and stock size for glass, yellow, and silver eels for the entire European stock, as well as the Bay of Biscay. The author noted that, because of the lack of input data, the assessment was not much more than a 'back of the envelope' calculation. The model provided estimates of 2000 million glass eels recruiting annually to Europe, most to the Bay of Biscay, and fishing mortality rate over their lifetime was estimated at 96–99%.

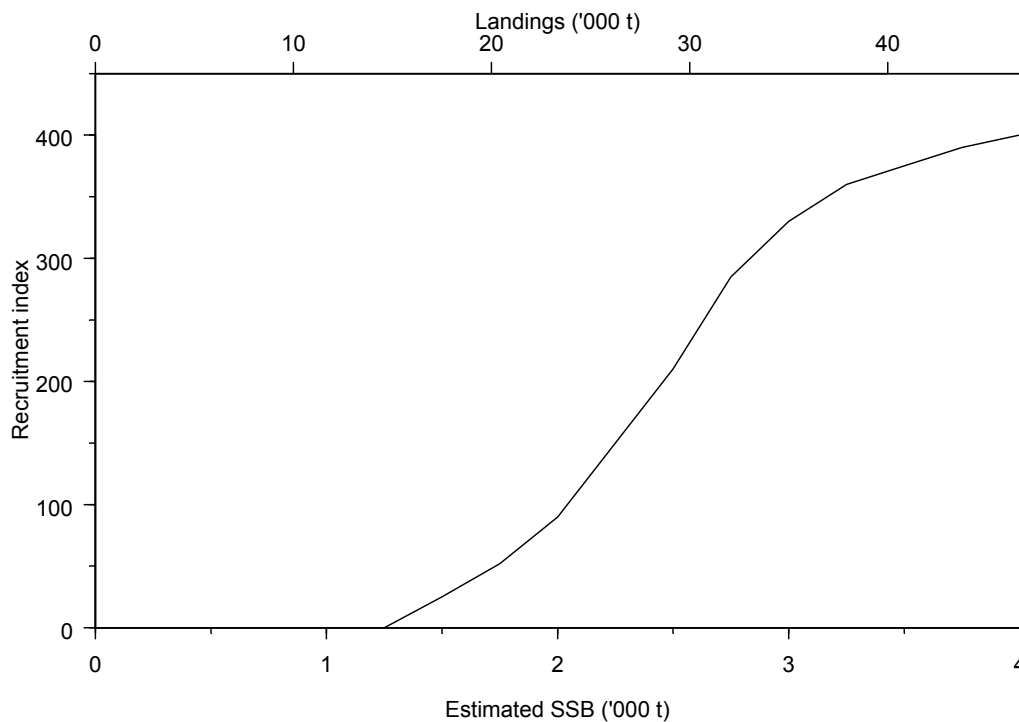
Dekker (2000a) used a Markov-chain matrix model structured by length cohort to assess the yellow eel fishery in Lake IJsselmeer, the Netherlands, and its impact on spawner production. He found that fishing on yellow eels reduced spawner escapement in males to 14% of that of an unexploited



population and for females to 0.14%. While there is no definitive stock-recruitment relationship available for European eels, Dekker (2000a) suggested that where fisheries exist, these could have had a significant impact on spawner escapement, and have contributed to the collapse of the European eel fishery.

A linear regression model of European eel landings that included explanatory variables such as latitude, country, and year was used by Dekker (2003a) to quantify trends in catches. He concluded that the results showed a trend of declining landings and suggested that insufficient spawning stock may have contributed to the decline in recruitment.

The 2005 report of the ICES/EIFAC working group on eels (WGEEL) (International Council for the Exploration of the Sea 2005) outlined the status of this species and highlighted information/data deficiencies required to produce a useful stock assessment model for the entire European stock, while also noting that the fishery was considered to be outside safe biological limits. The available data included landings, recruitment indices from discrete sites, and CPUE of yellow eels. A tentative stock-recruitment relationship based on landings (assumed to be proportional to spawning biomass) was postulated. The shape of the curve was sigmoidal and indicated that spawning stock biomass may have declined below a sustainable threshold (Figure 2).



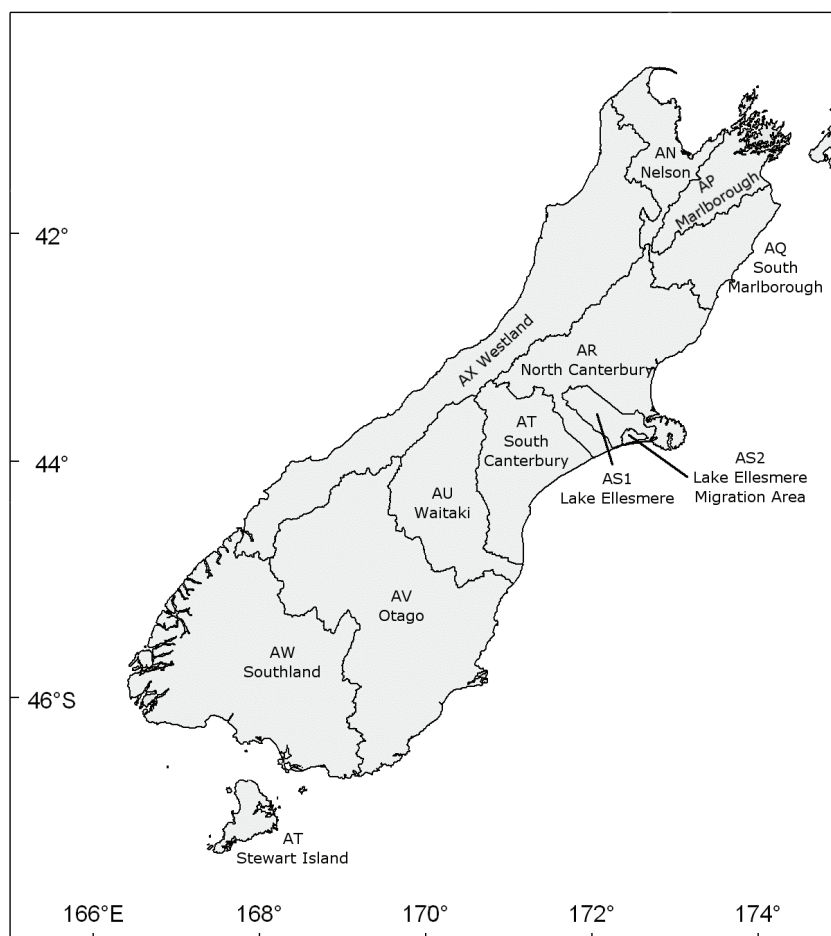
**Figure 2: Estimated stock-recruitment relationship for the European eel *A. anguilla* (reproduced from figure 3.2.1 in International Council for the Exploration of the Sea 2005). Spawning stock biomass (SSB) was assumed proportional to landings from the continental stock. (Note, only the trend line is shown, not the individual SSB-recruitment points from the original figure.)**

The stock-recruitment relationship was thought to exhibit signs of depensation or ‘Allee effect’, where at very low population levels negative feedback moves the stock towards extinction, and may occur if there were insufficient eels arriving at spawning grounds to successfully spawn, and hence lead to recruitment collapse. The report expressed concern that the spawning stock biomass of European *A. anguilla* might have been reduced to such a low level that recruitment is no longer sufficient and the population may have already collapsed. There are others, however, who consider that the steepness of the decline in the stock cannot be ascribed exclusively to the effects of fishing, habitat loss, and other anthropogenic impacts, and that changes to oceanic and climate may have been contributing factors (Knights 2003).

## 2. POPULATION MODEL METHODS

### 2.1 Introduction

Data suitable for the modelling of New Zealand longfin eel populations are limited, with the most suitable data being for the Southland longfin eel Fishstock<sup>2</sup> ANG15 (Otago/Southland, Figure 3). We summarise the available data for the Southland longfin eels, and outline a maturity-, sex-, and age-structured population dynamics model for this Fishstock. As the Southland longfin eel Fishstock has the largest set of data on age, length, and relative abundance, we restrict the investigation of a longfin eel model to this area alone. However, later we discuss how this analysis may be improved, and perhaps extended to cover longfin data from other areas.



**Figure 3: Map of the South Island showing the Eel Statistical Areas. (Note: ESA AW (Southland) was formerly known as ESA 20.)**

We implement the stock assessment model using a modified version of CASAL (Bull et al. 2005), where the modifications include (i) an option that defines the spawning stock biomass as calculated from either the biomass of both sexes combined or biomass of females only, and (ii) an option to include semelparous mortality (i.e., where all mature spawning individuals are removed from the model partition) immediately following spawning and the calculation of spawning stock biomass.

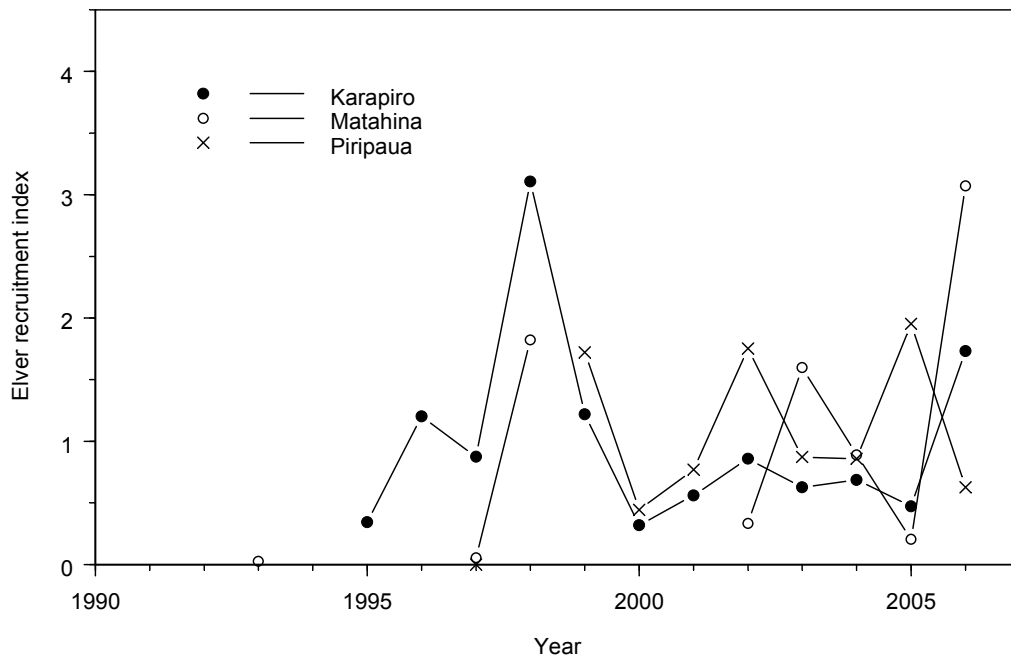
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<sup>2</sup> The term ‘Fishstock’ is used in this document to refer to the Ministry of Fisheries management units as defined under the QMS. The term ‘stock’ or ‘fish stock’ is used to denote a group of individuals who form a single discrete biological production unit.

## 2.2 Biological inputs, priors, and assumptions

### 2.2.1 Recruitment

Annual elver estimates were available for Karapiro, Matahina, and Piripaua from 1993 to 2006. Estimates were converted to an annual index by dividing each series by its mean (see Figure 4), and included within the model as observations of relative indices of numbers of eels aged 1–3, with a selectivity assumed to be 1.0 for eels aged 1, 2, and 3.

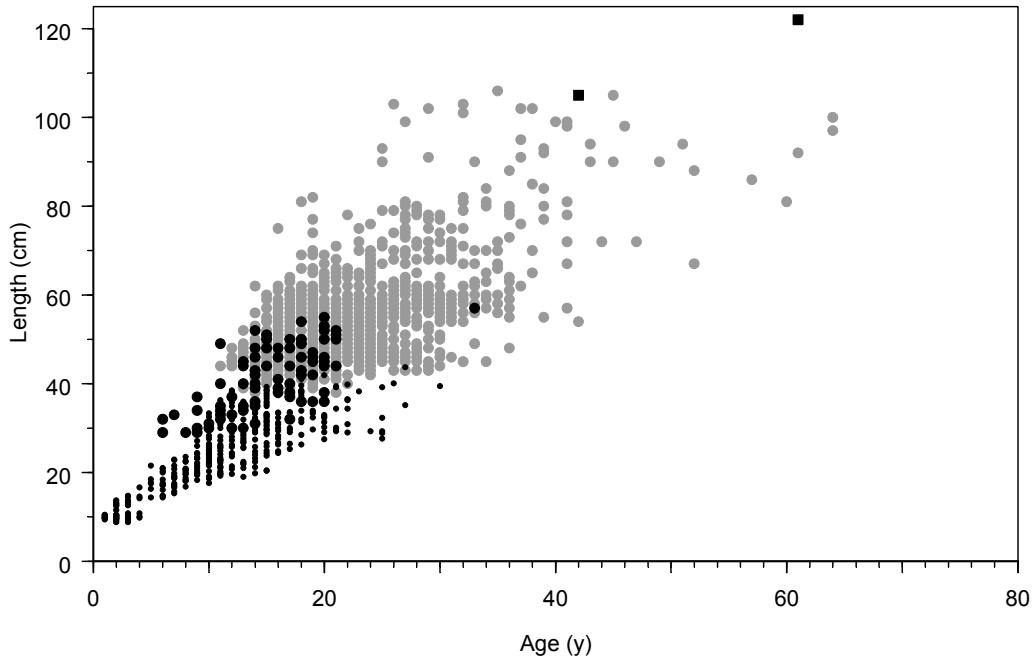


**Figure 4: Relative indices of recruitment for Karapiro (filled circles), Matahina (open circles), and Piripaua (cross) 1993–2006.**

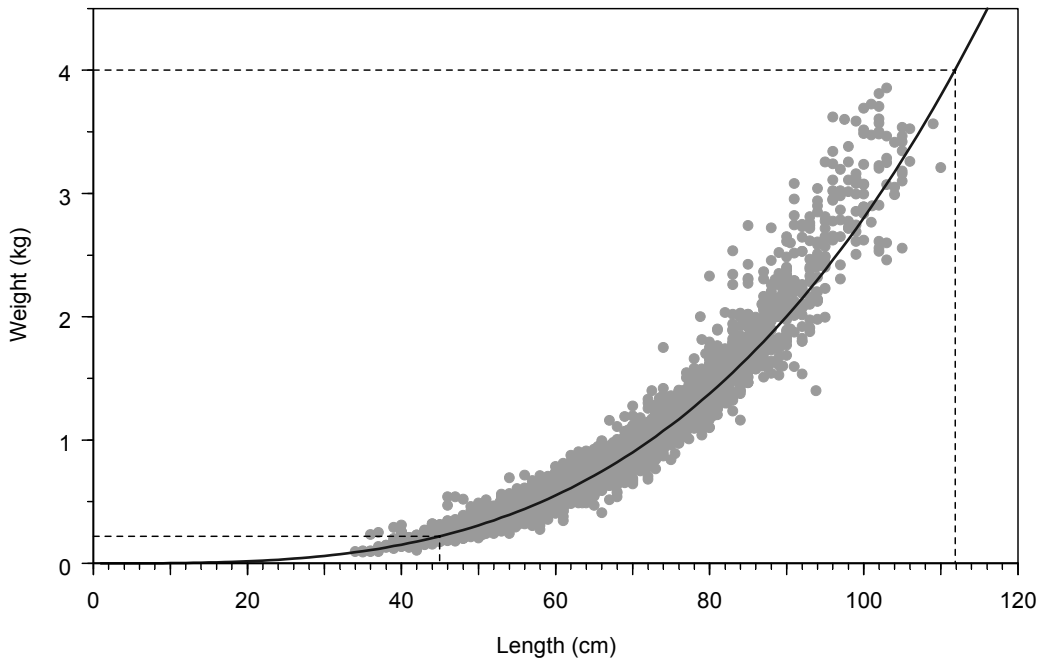
### 2.2.2 Growth

Horn (1996) reviewed the age and growth estimates for New Zealand freshwater eels. He reported that their growth was highly variable and probably dependent on food availability, water temperature, and eel density, and noted longfin eels were generally long lived with a maximum recorded age of 106 years. Beentjes & Chisnall (1998) reported growth rates from commercial catch sampling programme data (for 1995 to 1997) and proposed that growth rates were highly variable within and between catchments. Longfin eels took, on average, 17.5 years (range 12.2–28.7 years) to reach 220 g (minimum legal weight). The growth rates were likely to be linear, and there appeared little difference in growth rates between the sexes.

Southland catch sampling length-at-age data for longfin eels were available for 1996 ( $n=467$ ), 1997 ( $n=268$ ), and 1998 ( $n=364$ ) (Beentjes 1999, Beentjes & Chisnall 1997, 1998); for 1998 from experiments in transferring eels from the lower Clutha River to Lake Hawea (Beentjes 1998) ( $n=96$ ), large eels caught in Lake Hawea ( $n=2$ ) during that experiment, and from various locations throughout the Aparima River catchment ( $n=384$ ) (Graynoth 1999, McCleave & Jellyman 2004). The age data are shown in Figure 5. Data for length at age for individual eels were included within the model as observations of length at age, with the assumption that the samples were randomly selected from the population, conditional on length.



**Figure 5: Observations of longfin eel length and age from the Southland catch sampling data, 1996, 1997, & 1998 (gray circles), sampled from the lower Clutha River in 1998 (black circles), from Lake Hawea in 1998 (black squares), and the Aparima River (dots).**



**Figure 6: Observations of longfin eel length and weight (points) from the Southland commercial catch sampling data, 1996–2004, and the estimated length-weight relationship (line). Dashed lines indicate weights and approximate lengths equivalent to the minimum and maximum legal weight (220 g and 4000 g respectively).**

### 2.2.3 Maturity

The Ministry of Fisheries Plenary Report (Ministry of Fisheries Science Group 2006) reported that longfin eel migration appeared to be dependent on attaining a certain length/weight combination and condition, and the range in recorded age and length at migration for longfin eels was 11–34 years and

24–67 cm for males, and 27–61 years and 90–158 cm for females. That report implied (from the way the text was constructed) that the reference for these data is a review by Horn (1996). The basis for these numbers is not known, but does not appear to agree with either Horn (1996) or Todd (1980).

Horn (1996) reported that longfin males consistently migrated at a younger age (and smaller size) than females, with average ages at migration for males and females, respectively, 25 and 36 years. Todd (1980) reported migration rates for longfin eels, and noted that the age at migration varied between areas, possibly depending on local growth rates, with males maturing at a smaller length than females. Migration appeared to be dependent on attaining a certain length/weight combination and condition (Todd 1980), and the range in recorded age and length at migration for longfin eels was 56–73 cm for males (no age range given), and 25–60 years and 74–156 cm for females from Lake Ellesmere (Table 1). More recently, Beentjes & Chisnall (1998) and Beentjes (1999, 2005) estimated the length of migrating eels from commercial catch sampling in Southland (Table 1), and found that males migrated at mean lengths of between 52 and 65 cm for males and 87–94 for females.

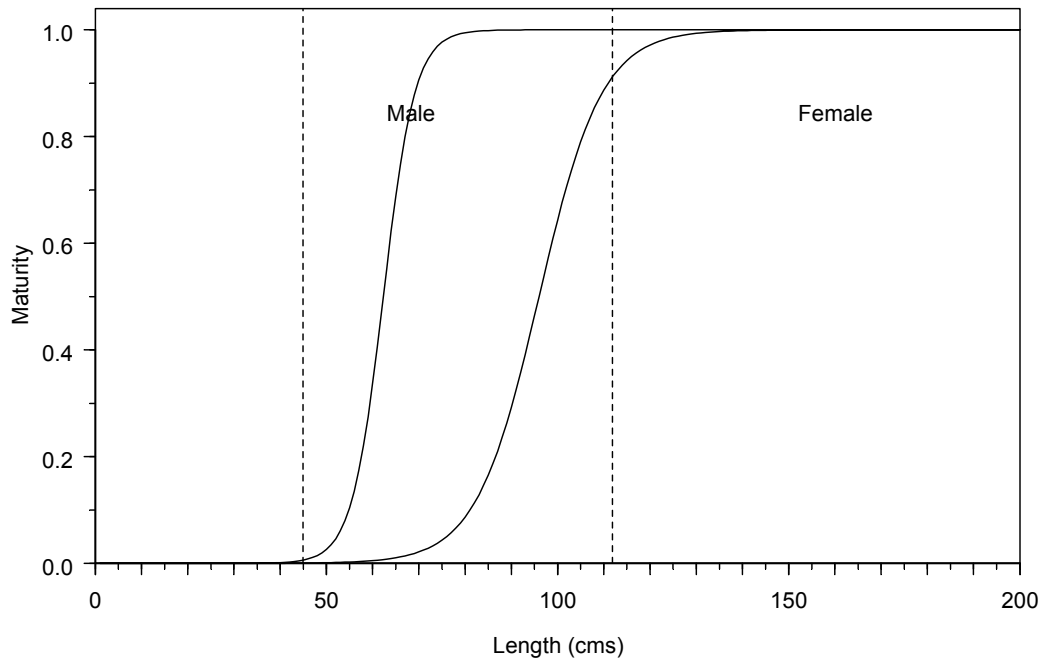
We assume a length-based maturation ogive (i.e., a logistic ogive that describes the proportions maturing in any year) for Southland longfin eels, using a logistic ogive, with parameter values based on a simple arithmetic average of those reported by Todd (1980), Beentjes & Chisnall (1998), and Beentjes (1999, 2005). The logistic ogive was parameterised using two parameters  $a_{50}$  (the value of 50% maturity), and  $a_{1095}$  (the value that describes the difference between 50% maturity and 95% maturity), i.e.,

$$f(x) = 1 / \left[ 1 + 19^{(a_{50}-x)/a_{1095}} \right]$$

with the parameters for males and females of  $a_{50} = 62.3$  cm and 96.0 cm, and  $a_{1095} = 10$  cm and 20.0 cm for males and females respectively (see Table 4), although we note that these estimates are somewhat arbitrary. The assumed maturity ogives for male and female longfin eels are shown in Figure 7.

**Table 1: Estimates of maturity from Lake Ellesmere (Todd 1980) and commercial catch sampling 1997, 1998, 2004 (Beentjes 1999, 2005, Beentjes & Chisnall 1998) (note: ‘n.a.’ indicates not reported).**

Data source	Male			Female		
	<i>n</i>	Mean length (s.d)	Range	<i>n</i>	Mean length (s.d)	Range
Lake Ellesmere	12	67 (6.55)	56–73	176	116 (17.4)	74–156
1997	11	65 (n.a.)	60–70	9	94 (n.a.)	80–105
1998	23	65 (n.a.)	n.a.	12	87 (n.a.)	n.a.
2004	1	52 (n.a.)	n.a.	12	87 (n.a.)	n.a.
Average		62.3			96.0	



**Figure 7: Assumed length-based maturity ogive (i.e., proportions maturing by length) for males and females. Vertical dashed lines indicate lengths equivalent to the minimum and maximum legal weight (220 g and 4000 g respectively).**

#### 2.2.4 Natural mortality

Jellyman (1995) estimated natural mortality rates from a sample of 146 unexploited longfin eels, ranging in length from 23.1 to 120.3 cm from Lake Rotoiti, Nelson Lakes National Park, New Zealand. Using 114 otoliths, he reported a maximum age of 106 years and a natural mortality rate of  $0.042 \text{ y}^{-1}$ . In these models we assume a constant natural mortality rate of  $0.042 \text{ y}^{-1}$  for Southland longfin eels.

As eels are believed to be semelparous (i.e., organisms that reproduce just once during their lifetime, after which they die), we assume 100% mortality of spawning eels immediately after spawning.

#### 2.2.5 Other sources of mortality

Beentjes et al. (2005) reported that eels are subject to significant sources of mortality from non-fishing activities, although levels have not been quantified. For example, they noted that direct mortality can occur through the mechanical clearance of drainage channels and damage by hydro-electric turbines and flood control pumping. In addition they noted that eel populations are likely to have been significantly reduced since European settlement from the 1840s by wetland drainage (wetland areas have been reduced by up to 90% in some areas), and habitat modification brought about by irrigation, channelisation of rivers and streams, and the reduction in littoral habitat. Ongoing drain maintenance activities by mechanical means to remove weeds may cause direct mortality through physical damage or by stranding and subsequent desiccation. In addition, migrating eels that are above dams are believed to not survive passage through hydro-electric generating turbines.

Estimates of the level of mortality from these sources are not known, and we ignore the effects of indirect mortality, habitat, and carrying capacity change in these models.

## 2.3 Commercial catch data

### 2.3.1 Minimum and maximum legal size

The minimum legal weight of longfin eels is defined as a weight of greater or equal to 220 g. In 1996, a maximum weight limit of 4000 g was introduced for the Fishstocks within the South Island. We assume that these limits are length based, and hence convert weight limits to length limits using the length-weight relationship above (e.g., 45 cm and 112 cm for the 220 g and 4000 g weight limits respectively).

### 2.3.2 Catch histories

Commercial catch histories for longfin eels for the Southland region have not previously been estimated and customary and recreational take are poorly known.

We estimated the commercial catch of longfin eels in Southland by assuming a constant proportion of total recorded catch of either the New Zealand or South Island eel (longfin and shortfin) catch for years where better estimates were not available. Here, commercial catch history for longfin eels in Southland (Eel Statistical Area ESA20/AW) was derived using, as a starting point, the landings for New Zealand for the period 1965 to 2005, published in the 2006 Plenary Report (Ministry of Fisheries Science Group 2006). The following data were used: 1965–1988 (2006 Plenary Report table 1, FSU and LFRR data), 1989 to 2005 (2006 Plenary Report table 2, LFRR, QMR, and MHR data), and 1992 to 2005 (2006 Plenary Report table 3, landings by Island from processors and LFRR/QMR data).

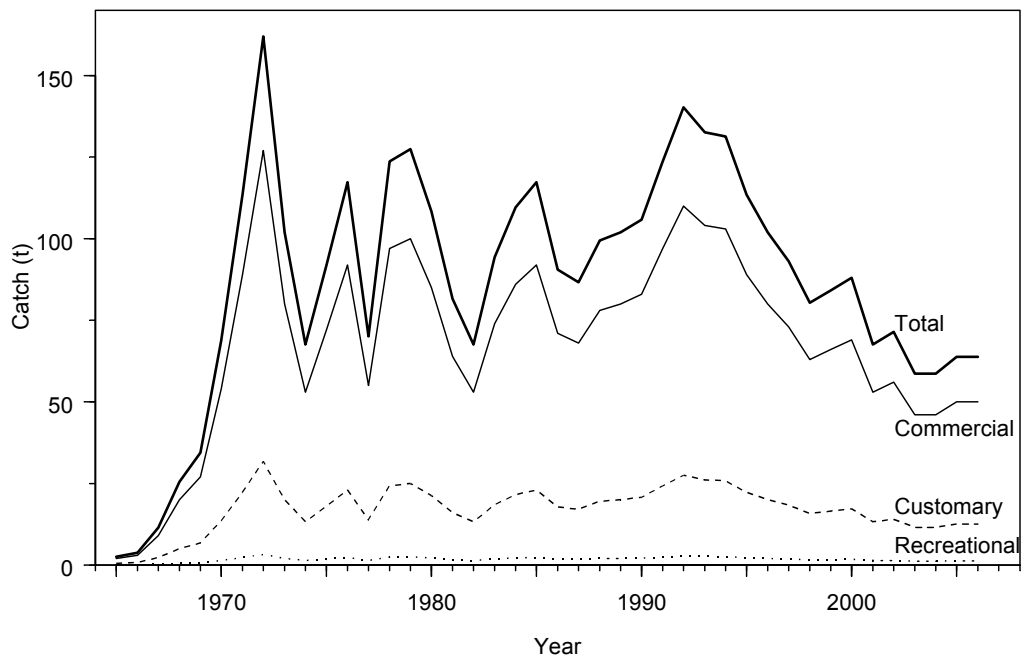
As there are no landed catch data by eel species (i.e., longfin versus shortfin) available for Southland, we estimated the catch of longfin eels from this area over the same time period (1965 to 2004–05) from catch effort data using estimated catch by species and statistical area. We used existing groomed datasets covering 1991 to 1999 from a previous CPUE analysis (Beentjes & Bull 2002).

The conversion of the New Zealand wide catch data from 1965 to 2005 to that for Southland longfin eels involved three steps.

- (i) Estimation of the South Island commercial catch. For 1992 to 2005, we used the values in table 3 of the 2006 Plenary Report. However, before 1992 there is no breakdown of catch by Island, hence we applied the mean proportion of the catch from the South Island (data from the 2006 Plenary Report, Table 3) from 1992 to 2005 ( $= 0.35$ ) to the catch totals from 1965 to 1991 to estimate South Island catch for the earlier period.
- (ii) Estimation of the Southland commercial catch. The estimated South Island catch was then multiplied by the mean proportion ( $= 0.21$ ) of the South Island catch of both species (CELR data 1991 to 1999) that was from reported to be from Southland.
- (iii) Estimation of the Southland longfin eel commercial catch. The estimated catch from Southland was then multiplied by the mean proportion ( $= 0.83$ ) of longfin in catches from Southland (CELR data 1991 to 1999) to give an estimate of Southland longfin catch.

The 2006 Plenary Report (Ministry of Fisheries Science Group 2006) reported that the allowance for customary and recreational catch in ANG15 for 2006 was 30.17 t and 3.17 t, about 25% and 3% respectively of the Total Allowable Commercial Catch (TACC) for shortfin and longfin eels combined (117.70 t). Without more complete information on the levels of customary and recreational catch, we assume that the historical levels of customary and recreational longfin eel catch were 25% and 3% of the longfin commercial catch in any year.

The estimated longfin catch for Southland, assumed in these models, is shown in Figure 8. Note, catch data for 2006 were unavailable at the time of writing this report, and we assumed that catches for 2006 were equal to those for 2005.



**Figure 8: Total (solid bold line), commercial (solid line), customary (dashed line), and recreational (dotted line) catches assumed for Southland longfin eels, 1965–2006. (Note, data for 2006 assumed to be equal to those for 2005.)**

### 2.3.3 Commercial catch sampling data

Commercial catch sampling data (Beentjes 1999, 2005, Beentjes & Chisnall 1997, 1998) for Southland were analysed to generate annual estimates of the length-frequency of the commercial catch. These data were samples taken from the commercial catch from the more heavily fished Oreti, Aparima, Waiau, Mataura, and Clutha Rivers only.

Estimates of the catch-at-length frequencies (with associated c.v.s) of the commercial catch were derived using catch-at-age software (Bull & Dunn 2002), using 1 cm length classes (30–110 cm), and ignoring sex. The software scaled the length frequency from each landing up to the landed catch, to yield length frequencies by landing and overall. The c.v.s were calculated by bootstrapping, using 300 bootstraps. The estimated catches-at-length for 1996, 1997, 1998, and 2004 are shown in Figure 9.

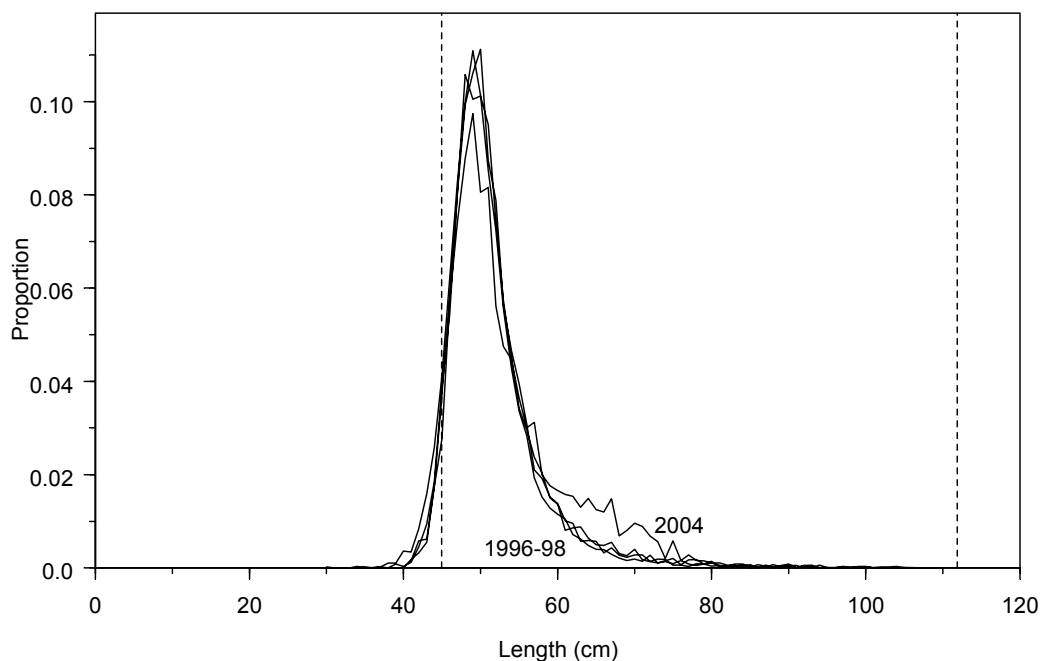
Mossburn Enterprises Ltd has processed most of the commercial catch of eels in the South Island, largely sourced from Otago and Southland. Their data provide the most comprehensive time series of weight grade and species composition for the South Island commercial eel fishery. The same weight grades, recorded in imperial units (lbs), have been used since the mid 1970s when records began. Mossburn Enterprises Ltd. provided the proportion (by weight) of eels within seven (for 1975–1995) or five (for 1996–2006) weight grades. The weight grades used between 1975–1995 were 0–1 lbs, 1–2 lbs, 2–3 lbs, 3–4 lbs, 4–5 lbs, 5–7 lbs, and over 7 lbs. After 1995, the last two grades were dropped, and the 5<sup>th</sup> grade became over 4 lbs.



As these were data that described proportions-by-weight for longfin eels, rather than the proportion-by-number, we first converted the proportions-by-weight into proportions-by-number by assuming the mean weight of eels within each weight class. Secondly, we used the length-weight relationship to convert the weight classes into length classes.

The mean weight of eels within each weight (or length) class was estimated by running a naive population model, assuming  $M = 0.042 \text{ y}^{-1}$  and estimating growth, and determining the proportions-at-length in an unexploited population. These were used to determine the mean length, and hence weight, of eels within a length (and hence weight) class. While this conversion is not ideal (i.e., it does not take into account changes within weight class resulting from exploitation), these observations cannot be included within the CASAL model (Bull et al. 2005) without substantive modifications<sup>3</sup>. The resulting proportions-at-length are shown in Figure 10.

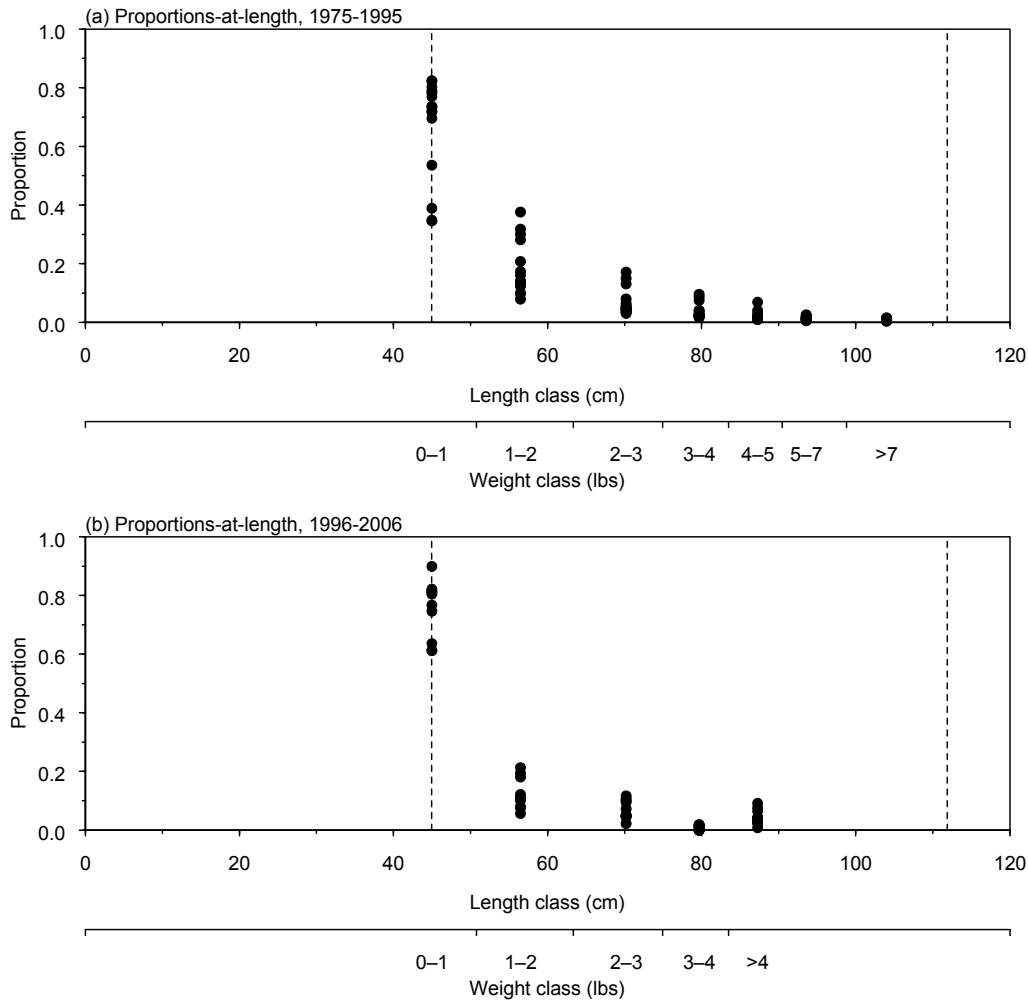
Both the commercial catch sampling data and the converted proportions-at-weight data were included within the model as observations of the commercial catch-at-length.



**Figure 9: Proportions-at-length from the catch sampling programme in 1996, 1997, 1998, and 2004 (Beentjes 1999, 2005, Beentjes & Chisnall 1997, 1998). Vertical dashed lines indicate lengths equivalent to the minimum and maximum legal weight (220 g and 4000 g respectively). (See later for plots of the annual distributions.)**

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<sup>3</sup> While these modification would be relatively easy to implement, they would require a significant amount of time to implement.



**Figure 10: Proportions-at-length for 1975–2006, as derived from the proportions-at-weight data supplied by Mossburn Enterprises Ltd. Black dots represent individual years. The last (right-most) points in each graph represent plus groups. Vertical dashed lines indicate lengths equivalent to the minimum and maximum legal weight (220 g and 4000 g respectively). (See later for plots of the annual distributions.)**

### 2.3.4 Catch-effort data

Beentjes & Dunn (2003a) updated CPUE indices for Southland longfin eels (Eel Statistical Area ESA20/AW) using data up to the end of the 2001. The indices are reproduced in Table 2. CPUE indices were included within the model as mid-season observations of the relative abundance of the commercially selected population.

## 2.4 Other data

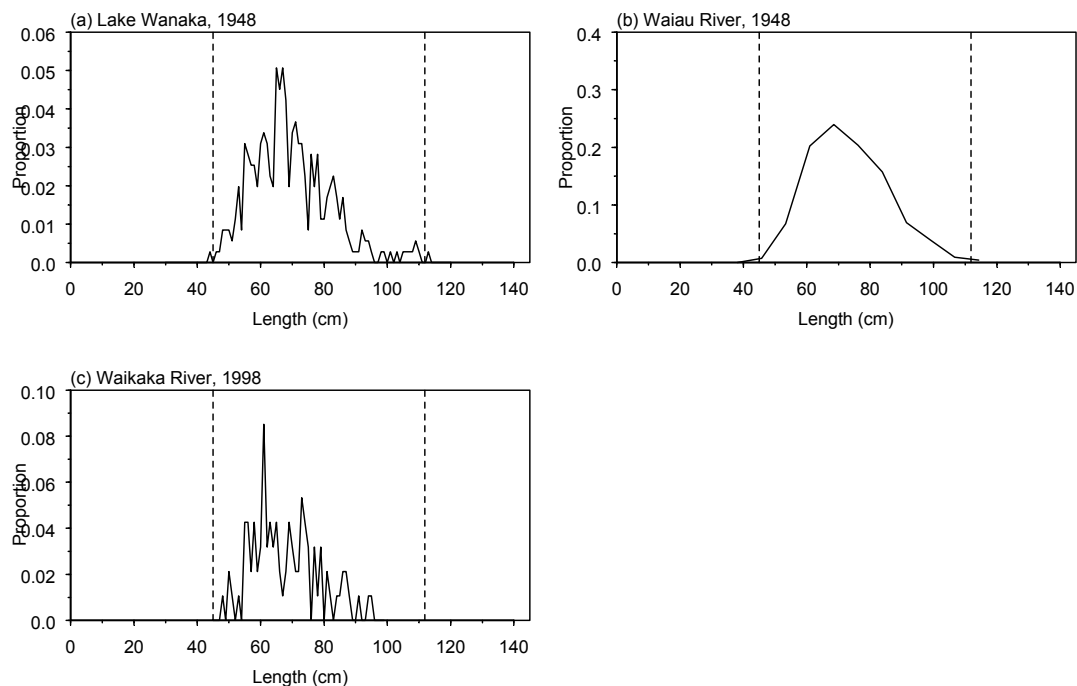
There are a number of length frequency data for areas that are believed to be either unexploited or were observed before exploitation had begun. The data are described below and shown in Figure 11.

Burnet (1952) reported on the length frequency of longfin eels in Lake Wanaka ( $n = 618$ ) and the Waiau River ( $n = 534$ ) for 1948 from sampling experiments. Beentjes (1999) collected length frequency from catch sampling from Waikaka River, an area that was believed to be previously unfished (or at least very lightly exploited) ( $n = 94$ ).

**Table 2: CPUE indices, upper and lower 95% confidence intervals, standard errors, and c.v.s for ESA20/AW, reported by Beentjes & Dunn (2003a).**

Year	Index	Lower C.I.	Upper C.I.	s.e.	c.v.
1991	1.550	1.411	1.704	0.047	0.047
1992	1.227	1.131	1.331	0.041	0.041
1993	1.190	1.095	1.295	0.042	0.042
1994	1.252	1.141	1.375	0.047	0.047
1995	1.022	0.929	1.125	0.048	0.048
1996	1.030	0.918	1.155	0.057	0.057
1997	0.950	0.854	1.056	0.053	0.053
1998	0.848	0.766	0.938	0.050	0.051
1999	0.693	0.623	0.770	0.053	0.053
2000	0.880	0.799	0.969	0.048	0.048
2001	0.682	0.586	0.794	0.076	0.076

For the single-area models, we include these data as observations within the model as an estimate of the pre-exploitation length frequency, and associate them with the first year of the model (1965). In the two-area models, the data from Burnet (1952) are included as pre-exploitation length frequencies as above, but data for the Waikaka River were included as estimates of the protected area length frequencies in 1998.



**Figure 11: Proportions-at-length of longfin eels for (a) Lake Wanaka in 1948 (Burnet 1952), (b) Waiau River in 1948 (Burnet 1952), and (c) Waikaka River in 1998 (Beentjes 1999). Vertical dashed lines indicate lengths equivalent to the minimum and maximum legal weight (220 g and 4000 g respectively).**

## 2.5 Population dynamics

The initial models were sex-, maturity-, and age- structured, with ages from 1 to 100 years, with the last age group a plus group (i.e., an aggregate of all eels aged 100 and older). The annual cycle was broken into three discrete time steps (Table 3), described below.

The models were run from 1965 to 2006, and were initialised assuming an equilibrium age structure at a pre-exploitation equilibrium biomass, i.e., a constant recruitment assumption. Within each model the

commercial, customary, and recreational fisheries were implemented as three simultaneous fisheries, with an equal and constant selectivity parameterised by a length-based logistic ogive. Two alternative model structures were investigated. The first considered the entire population to be available to the fisheries (the single-area model), while the second allowed a fixed proportion of the stock to recruit to a ‘protected area’ that was not subject to any fishing mortality (the two-area model). The two scenarios are graphically shown in Figure 12. The two-area model structure differed from the single-area model in that it was assumed a fixed proportion of recruits are unavailable to fishing, i.e., they recruit to and spend their entire lives up to the point of spawning within a protected area. This was carried out within the models by recruiting all fish at age 1 to a ‘protected area’, then, in the following time step, migrating a fixed proportion of these fish to the ‘fished’ area. At maturation, all mature fish (from both areas) were migrated to a third area, the spawning area, after which spawning stock biomass was calculated and semelparous mortality applied.

Recruitment was assumed to occur at the beginning of the second time step. Recruitment was assumed to be 50:50 male to female, and was parameterised as a year class strength multiplier (assumed to be equal to one for 1964–2005), multiplied by an average (pre-exploitation) recruitment ( $R_0$ ) and a spawning stock-recruitment relationship. While there are some suggestions that eel stocks may have a sigmoidal stock recruitment relationship (i.e., depensation — see International Council for the Exploration of the Sea 2005), we assume a Beverton-Holt relationship with steepness  $h = 0.75$  (Table 4), but we also investigate the effect of an assumption of no stock-recruit relationship.

The value of  $h = 0.75$  was based on meta-analyses of Myers et al. (1999) and Punt et al. (2005). Table 1 in Myers et al. (1999) summarised estimates of  $h$  for a wide variety of international fisheries. We used these values to derive plausible values of  $h$  for individual species selected from time series of greater than 10 years from the orders Gadiformes (including cods and hakes), Lophiiformes, Perciformes (Lutjanidae and Percichthyidae, Sparidae, and Xiphiidae only), Pleuronectiformes (halibut only), Salmoniformes (pike only), Scorpaenidae, and *Anoplopoma fimbria* (sablefish). From these data, a median value of  $h = 0.73$  (90% range 0.32–0.94) was derived. In addition, Punt et al. (2005) analysed stock and recruitment data to estimate priors for the steepness of the stock-recruitment relationship. Their recommendations were based on the median posteriors of estimated values of steepness. They recommended that “... if a single default point estimate of steepness be used in a stock assessment, then the default should be  $h = 0.907$  for Clupeiformes, Gadiformes, and Pleuronectiformes, and  $h = 0.757$  for other species” (i.e., Aulopiformes, Perciformes, Salmoniformes, and Scorpaeniformes). We note that alternative stock-recruitment relationships are plausible (and perhaps meta-analyses of salmon fisheries may provide a better initial value), and should be investigated in future work. In projections we assume future recruitment variability  $\sigma_R = 0.6$ , the value proposed by Punt et al. (2005) as a suitable default value for  $\sigma_R$  in cases where better information was not available.

Maturation was assumed to occur in the second time step, with males and females becoming mature with rates described by a time-invariant length-based logistic ogive (see Figure 7). We assume that all mature eels migrate to spawn, and that estimates of migration rates by length are the same as the rates of maturation by length. Two approaches to calculating spawning stock biomass (SSB) were considered. The first calculated the mid-season SSB as the sum of the biomass of male and female eels that were mature. The second scenario calculated SSB as the sum of the biomass of female eels only. In both cases, spawning stock biomass was evaluated in the second time step, after half of that time step’s natural mortality. Following spawning and the calculation of SSB, all mature eels were assumed to die (i.e., assumed to be semelparous). The parameters ( $a_{50}$  and  $a_{t095}$ ) for maturity are given in Table 4.

Fishery selectivities were assumed to be length based, and equal (by length) for males and females, using a logistic selectivity ogive, also parameterised using estimable parameters  $a_{50}$  and  $a_{t095}$ .

Natural mortality was assumed to be constant across age and sex classes, using  $M = 0.042 \text{ y}^{-1}$  (Jellyman 1995), see Table 4. Fishing mortality was applied only in the first time step and consisted of

the sum of the commercial, customary, and recreational catch, where both recreational and customary catch levels were assumed to be a fixed proportion of the commercial catch (described earlier). Catches for 2006 were not available in time for this report, but were assumed to be equal to the 2005 catch. The process applied in the model was to remove half of the natural mortality occurring in that time step, then apply the mortality from the fisheries instantaneously, then to remove the remaining half of the natural mortality, using the catch equation, i.e.,

$$U_f = \frac{C_f}{\sum_{ij} \bar{w}_{ij} S_{ij} n_{ij} \exp(-0.5tM_{ij})}$$

where  $C_f$  is the catch for fishery  $f$ ,  $S_{ij}$  is the selectivity at age  $i$  and sex  $j$ ,  $\bar{w}_{ij}$  is the mean weight of eels of age  $i$  and sex  $j$ ,  $n_{ij}$  is the number of fish,  $M_{ij}$  is the natural mortality,  $t$  is the proportion of the year's natural mortality in the time step, and  $U_f$  is the resulting exploitation rate. The maximum possible exploitation rate associated with  $f$  was defined as the maximum proportion of eels taken from any age/sex class in the area affected by fishery  $f$  (and constrained to be less than or equal to 0.4), i.e.,

$$\hat{U}_{\max}(f) = \max_{i,j} (S_{ij} U_f)$$

In the age incrementation time step all eels age by 1 year, with the exception of eels in the plus group — these become the sum of eels aged 99 years and those aged over 99 years.

Length-weight parameters are described in Table 4. Growth curves for eels are assumed to be linear, and modelled using the Schnute curve (Schnute 1981), where mean length at age ( $l$ ) was defined as,

$$l = y_1 + (y_2 - y_1) \frac{\text{age} - \tau_1}{\tau_2 - \tau_1},$$

with parameters  $y_1$  and  $y_2$  estimated within the model with  $\tau_1 = 1$  and  $\tau_2 = 100$ .

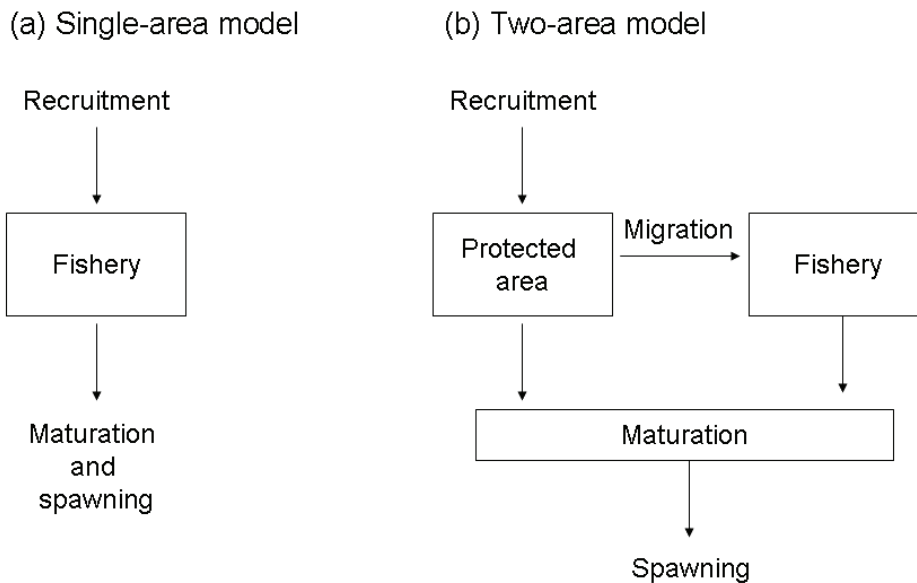
**Table 3: Annual cycle of the stock model, showing the processes taking place at each time step, their sequence within each time step, and the available observations. Fishing and natural mortality that occur within a time step occur after all other processes, with half of the natural mortality for that time step occurring before and half after the fishing mortality.**

Step	Processes	$M^1$	Age <sup>2</sup>	Observations	
				Description	% $M^3$
1	Fishing	0.50	0.50	Catch-at-length	50
				CPUE	50
				Age-length	50
2	Recruitment Recruitment migration (in the two-area models) Maturation Spawning migration Spawning	0.50	0.50		
3	Increment age	0.00	0.00		

1.  $M$  is the proportion of natural mortality that was assumed to have occurred in that time step.
2. Age is the age fraction for determining length at age, that was assumed to occur in that time step.
3. % $M$  is the percentage of the natural mortality in each time step that was assumed to have taken place at the time each observation was made.

**Table 4: The biological parameters (natural mortality, growth, length-weight relationship, and length at maturity) assumed within the population models.**

Relationship	Parameter	Values		
		Male	Female	All
Natural mortality	$M$ ( $y^{-1}$ )			0.042
Length-weight	$a$ ( $kg.cm^{-1}$ )			$1.205 \times 10^{-6}$
	$b$			3.183
Length at maturity (cm)	$a_{50}$ ( $\pm a_{1095}$ )	62.3 ( $\pm 10.0$ )	96.0 ( $\pm 20.0$ )	
Stock-recruitment relationship	$h$			0.75



**Figure 12: The two population model structures assumed (a) the single-area model, and (b) the two-area model.**

### 2.5.1 Observation error assumptions and process error

Observations, described above, that were included within the model were (i) CPUE indices for 1991–2001, (ii) commercial catch-at-length observations from the catch sampling programme for 1996–1998 and 2004, (iii), commercial catch-at-length observations from Mossburn Enterprises Ltd. commercial grade data for 1975–1995 in seven weight categories and 1996–2006 in five weight categories, (iv) relative abundance indices for elver data from Karapiro, Matahina, and Piripaua for years between 1993 and 2006, (v) proportions-at-length from Lake Wanaka, Waiau River, and Waikaka River as an estimate of the pre-exploitation length frequency, and (vi) age-length observations from commercial catch sampling in 1995, 1996, 1997, and 1998, and also from biological sampling of the Aparima River in 1995 (NIWA, unpublished data).

In general, additional variance, assumed to arise from differences between model simplifications and real world variation, should be considered for all observations. Adding such additional errors to each observation type typically has two main effects; (i) it alters the relative weighting of each of the data sets (observations) used in the model, and (ii) it typically increases the overall uncertainty of the model, leading to wider credible bounds on the estimated and derived parameters. However, in developing these models we ignored data weightings and issues arising from incorrect relative weighting of the different data sets (other than additional process errors for the CPUE indices — see below). We note that consideration of relative data weighting and process error should be considered in future developments of these models.

## 2.5.2 CPUE indices

Standardised CPUE indices were assumed to be relative mid-season vulnerable biomass indices, with an associated catchability constant  $q$ . The catchability constant was assumed to be a nuisance parameter, i.e., the value of  $q$  was chosen that minimised the objective function, calculated algebraically (see Bull et al. (2005) for more detail). A lognormal likelihood was used for the CPUE indices, with observations  $O_i$ , c.v.  $c_i$ , and expected values  $qE_i$ , where

$$-\log(L) = \sum_{i=1}^n \left( \log(\sigma_i) + 0.5 \left( \frac{\log(O_i/qE_i)}{\sigma_i} + 0.5\sigma_i \right)^2 \right)$$

$$\text{and } \sigma_i = \sqrt{\log(1 + c_i^2)}$$

Additional process error was used in some runs, where the applied c.v.  $c'_i$  was determined from the process error  $c_{PE}$  (arbitrarily assumed to be 10% in some model runs, otherwise ignored) and the observed c.v.s  $c_i$  by,

$$c'_i = \sqrt{c_i^2 + c_{PE}^2}$$

## 2.5.3 Proportions-at-length

The proportions-at-length data were fitted to the modelled proportions-at-length composition using a multinomial likelihood, i.e.,

$$-\log(L) = -\log(N!) + \sum_i \left[ \log((NO_i)!) - NO_i \log(E_i) \right]$$

where  $O_i$  are the observed proportions-at-length  $i$ ,  $E_i$  are the model expected proportions-at-length  $i$ , and  $N$  is the effective sample size. The effective sample sizes assumed for the proportions-at-length data were assumed to be  $N = 50$  for all such observations.

## 2.5.4 Age-length observations

Age-length observations were included within the model using the random-at-size likelihood described by Bull et al. (2005), i.e., where,

$$L = \left[ \sum_{a'} N_{a's} M_{a'a} f_{a's}(l) \right] / \left[ \sum_{a's'} N_{a's'} f_{a's'}(l) \right]$$

where  $N_{a's}$  is the number of eels of true age  $a'$  and sex  $s$ ,  $M_{a'a}$  is the probability that an eel of true age  $a'$  is observed as age  $a$ , and  $f_{a's}(l)$  is the probability density function describing the distribution of sizes for a given (true) age  $a'$  and sex  $s$ . Ageing error was assumed using an ageing error misclassification matrix  $A$ , defined from a normal distribution with constant c.v. = 0.1.

## 2.6 Model estimation

The model parameters were estimated by minimising an objective function, which is the sum of the negative log-likelihoods from the data, negative-log priors (in a Bayesian analysis), and penalties that constrain the parameterisations, i.e., the objective function in a Bayesian analysis for  $\mathbf{p}$ , the vector of the free parameters,  $L$  the likelihood function, and  $O_i$  the  $i$ th observation was

$$\text{Objective}(\mathbf{p}) = -\sum_i \log[L(\mathbf{p} | O_i)] - \log[\theta(\mathbf{p})]$$

where  $\theta$  is the joint prior (and penalty) density of the parameters  $\mathbf{p}$ . The penalties and priors are described below.

Model fits were evaluated at the maximum of the posterior density (MPD) only. While Monte Carlo Markov Chain (MCMC) methods more appropriately describe model fits and uncertainty, we use the MPD as a proxy for these in developing the initial models. However, future development would need to consider MCMC methods.

## 2.7 Penalties

Two types of penalties were included within the model. First, the penalty on the catch constrained the model from returning parameter estimates where the population biomass was such that the catch from an individual year would exceed the maximum exploitation rate (0.4). Second, in models where relative year class strengths were estimated, the model included a penalty to encourage the mean year class strength to be one.

## 2.8 Priors

Priors were defined for all free parameters in the models. The free parameters, starting values for the minimisation, and bounds are given in Table 5. In models presented here, the priors that were generally chosen were relatively non-informative (i.e., uniform and log-uniform).

**Table 5: Number ( $N$ ), start values, priors, and bounds for the free parameters (when estimated) for the model runs.**

Parameter	$N$	Start value	Prior	$\mu$	c.v.	Bounds	
						Lower	Upper
$B_0$	1	500	Uniform-log	–	–	100	1500
CPUE $q$	1	–	Uniform	–	–	$1 \times 10^{-12}$	$1 \times 10^1$
Recruitment $q$	3	–	Uniform	–	–	$1 \times 10^{-12}$	$1 \times 10^1$
Selectivities	$a_{50}$	45.0	Uniform	–	–	5.0	100.0
	$a_{1095}$	4	Uniform	–	–	1.0	100.0
Length-at-age (growth)	$y_1$	1	Uniform	–	–	0	25
	$y_2$	1	Uniform	–	–	150	350
	c.v.	1	Uniform	–	–	0.0	1.0
Year class strength	$y_i$	36	Lognormal	1.0	1.1	0.01	100.0

## 2.9 Sensitivity runs

An initial, simple, model run was first defined, and then subsequent runs (described as sensitivities) were run on modifications to that initial run. The sensitivity runs are summarised in Table 6.



**Table 6: Labels and descriptions of the sensitivity runs.**

Model run	Description
1 Initial case	The initial model
2 Female SSB	The initial model, but calculating SSB from female biomass only
3 YCS	The female SSB model, but also estimating relative year class strength
4 CPUE c.v.	The female SSB model, but ignoring the additional CPUE process error
5 Two-area	The female SSB model, but modified to have two areas (protected and fishing areas)
6 Two-area, YCS	The two-area model, but also estimating relative year class strength
7 Two area, CPUE c.v.	The two-area model, but ignoring the additional CPUE process error
8 Two-area, no SR	The two-area model, but without a stock recruitment relationship

### 3. RESULTS

#### 3.1 Objective function values and biomass estimates

Objective function values for the MPD estimates for each of the model runs are given in Table 7 and estimates of initial ( $B_0$ ) and current biomass in Table 8. Comparison of the objective function values suggested that there were only minor differences in fit between each of the data sets between each of the model runs — except that a small improvement in fit was obtained to the CPUE indices and proportions-at-length data when either relative year class strengths were estimated or the additional CPUE process error ignored. Likelihood values for the Lake Wanaka, Waiiau River, and Waikaka River proportions-at-length and the age-length observations were very similar between runs, with estimated growth parameters also very similar between runs (see Table 9).

The initial model gave an estimate of the initial spawning stock biomass of 182 t, with current biomass about 21% of the pre-exploitation value. However, in this scenario, the current spawning stock biomass was predominantly made up of males. When the model was re-run using female biomass only as an index of spawning stock state (female SSB model), the current spawning stock biomass was estimated as about 5% of pre-exploitation levels (Figure 13).

In general, estimates of biomass from the two-area models were very similar to those for the single-area models if the proportion of the population in the protected area is ignored, i.e., the weight of eels within the fished areas and their absolute contribution to the total spawning stock biomass remained similar across all model runs. The models suggested that most of the current spawning stock biomass estimated within the two-area models was from the protected areas (Figure 13). Across all models, the estimated pre-exploitation spawning stock biomass ranged between about 120 and 180 t, with biomass of legal sized eels (defined as those with a length greater or equal to 45 cm) ranged from 1700 t to 2100 t, with the current biomass between 450 t and 1100 t (Table 8).

**Table 7: MPD objective function values and number of estimated parameters for the model runs.**

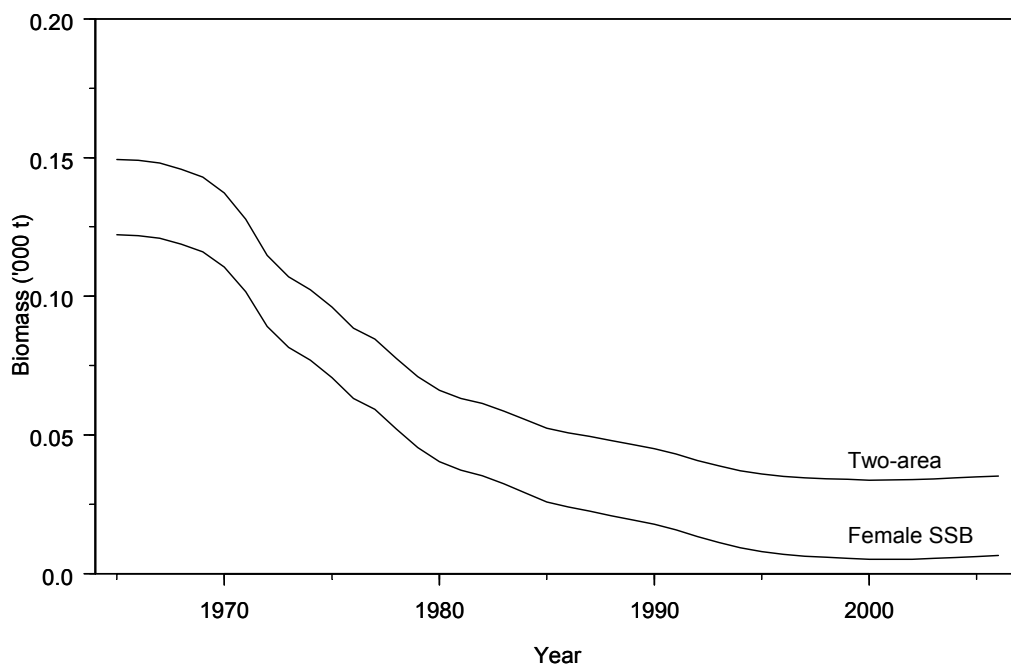
Objective function Component	Model run								
	Initial	Female SSB	YCS	CPUE	c.v.	Two-area	Two-area YCS	Two-area CPUE	Two-area no SR
CPUE	-17.6	-17.9	-20.2	-9.3	-17.6	-19.8	-7.2	-15.3	
Karapiro recruitment	-	-	32.1	-	-	28.5	-	-	
Matahina recruitment	-	-	13.6	-	-	8.0	-	-	
Piripaua recruitment	-	-	50.1	-	-	52.7	-	-	
Lake Wanaka LF	54.3	54.3	53.9	54.2	54.3	54.1	54.3	54.3	
Waiau River LF	13.2	13.2	13.0	13.2	13.3	13.1	13.3	13.3	
Waikaka River LF	53.8	53.8	53.6	53.8	53.8	53.5	53.8	53.8	
Catch-at-length sampling	201.2	202.9	204.5	201.1	201.3	203.3	199.9	201.2	
Mossburn LF (1975–1995)	368.5	374.7	367.0	373.7	368.5	363.0	367.7	368.9	
Mossburn LF (1996–2006)	303.0	306.2	300.0	307.6	303.0	300.2	304.1	303.5	
Age-length data (1996)	1 511.5	1 504.4	1 505.0	1 504.5	1 515.1	1 513.4	1 515.3	1 513.9	
Age-length data (1997)	831.8	832.7	832.9	833.8	833.8	832.9	834.6	833.6	
Age-length data (1998)	1 092.5	1 087.4	1 080.5	1 089.0	1 095.1	1 082.8	1 096.4	1 094.2	
Age-length data (Clutha)	272.6	269.1	281.5	269.1	275.6	284.0	275.7	275.0	
Age-length data (Hawea)	12.3	12.7	12.7	12.6	12.6	12.7	12.6	12.6	
Age-length data (Aparima)	1 012.3	1 019.1	969.1	1 019.2	1 006.2	961.2	1 005.8	1 007.8	
Sub-total (observations)	5 709.5	5 712.5	5 749.3	5 722.5	5 715.1	5 743.6	5 726.3	5 716.9	
Penalties	0.0	0.0	0.1	0.0	0.0	0.1	0.0	0.0	
Priors	-0.6	-1.1	-39.5	-1.1	-0.8	-40.2	-0.7	-0.8	
Total objective function	5 708.9	5 711.3	5 709.9	5 721.5	5 714.3	5 703.4	5 725.6	5 716.1	
Number of parameters	12	12	47	12	12	47	12	12	

**Table 8: Selected MPD parameter and derived parameter values — initial and current legal sized biomass (t) for the protected and fished areas for the two-area models, and equilibrium initial and current SSB (t) for all models.**

Model	Protected areas			Fished areas			SSB		
	Initial	Current	%	Initial	Current	%	$B_0$	$B_{2006}$	$B_{2006}(\%B_0)$
Initial case	–	–	–	1 731.0	574.2	33.2	182.2	39.0	21.4
Female SSB	–	–	–	1 763.4	531.1	30.1	122.4	6.5	5.3
YCS	–	–	–	1 738.7	450.3	25.9	119.8	6.6	5.3
CPUE c.v.	–	–	–	1 737.9	485.9	28.0	120.1	5.1	4.3
Two-area	429.2	403.4	94.0	1 715.5	566.3	33.0	149.5	35.2	23.6
Two-area, YCS	417.0	419.2	100.5	1 667.0	628.0	37.0	144.4	38.4	26.6
Two area, CPUE c.v.	425.1	399.4	93.9	1 699.4	538.6	31.7	147.8	34.1	23.1
Two-area, no SR	429.4	429.4	100.0	1 716.4	668.0	38.9	149.6	37.0	24.7

**Table 9: Selected MPD parameter values — Schnute linear growth parameters ( $y_1$ ,  $y_2$ , and c.v.), Lake Wanaka selectivity parameters ( $a_{50}$  and  $a_{t095}$ ), and the fishing selectivity parameters ( $a_{50}$  and  $a_{t095}$ ) for all models.**

Model	$B_0$	Growth			Wanaka selectivity		Waiiau Selectivity		Waikaka Selectivity		Fishing selectivity	
		$y_1$	$y_2$	c.v.	$a_{50}$	$a_{t095}$	$a_{50}$	$a_{t095}$	$a_{50}$	$a_{t095}$	$a_{50}$	$a_{t095}$
Initial case	182.2	6.4	200.9	0.16	66.8	4.1	78.6	4.0	64.1	4.5	52.6	1.0
Female SSB	122.4	6.4	203.7	0.16	66.6	4.1	79.2	4.1	64.7	4.3	51.1	1.0
YCS	119.8	6.7	204.1	0.15	65.8	1.9	78.1	2.7	62.8	2.2	51.2	1.0
CPUE c.v.	120.1	6.5	203.2	0.16	66.7	4.3	78.9	4.6	64.4	4.4	51.0	1.0
Two-area	149.5	5.8	203.4	0.16	66.6	4.1	78.7	4.0	64.0	4.5	52.6	1.0
Two-area, YCS	144.4	6.0	203.5	0.15	66.5	3.8	78.4	3.8	63.0	4.2	52.7	1.0
Two area, CPUE c.v.	147.8	5.8	203.0	0.16	66.8	4.2	78.7	4.1	64.2	4.5	52.5	1.0
Two-area, no SR	149.6	5.7	203.2	0.16	66.8	4.1	79.0	4.0	64.4	4.4	52.5	1.0



**Figure 13: Estimated MPD SSB trajectory for the female SSB and two-area models.**

### 3.2 Summary of model fits

Estimated parameter fits for all models are given in Table 9, and relative year class estimates for the YCS and two-area YCS model in Figure 14. Both series suggested a pattern of initially high recruitment followed by lower than average recruitment, and then about average recruitment in the most recent years. In general, model parameter estimates were similar across all models. Fishing selectivity ogives tended to have a sharp increase, effectively knife-edge at about 50 cm, while the Lake Wanaka, Waiiau River, and Waikaka River selectivities were usually smoother, with full selectivity at about 65–84 cm (Figure 15).

Model fits to the observed proportions-at-length data and age-length observations were similar for all models and, in general, were not ideal. The fits to the female SSB model for the Lake Wanaka, Waiiau River, Waikaka River proportions-at-length are shown in Figure 16, fits to the commercial catch sampling data in Figure 17, and the fits to the Mossburn Enterprises Ltd. proportions-at-length in Figures 18–19. Fits to the commercial catch sampling data often suggested a greater number of larger eels than was observed, while the reverse was the case for the Mossburn Enterprises Ltd proportions-

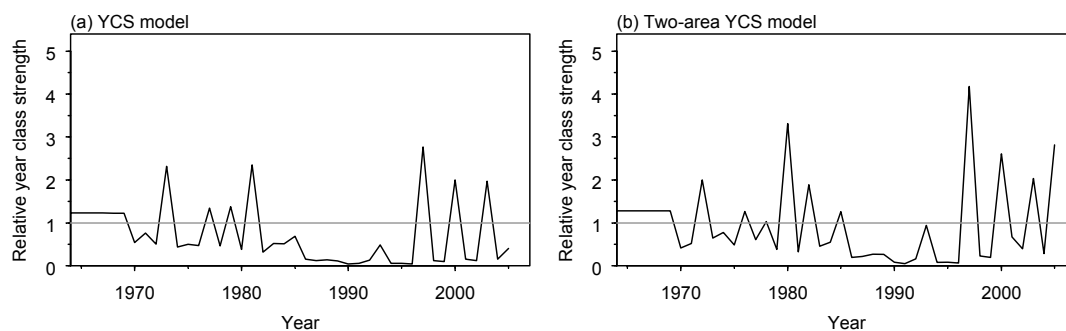
at-length. Most of the ‘peaks’ in the observed distributions were not well fitted, except in the first few years of the Mossburn Enterprises Ltd proportions-at-length data.

CPUE indices were less well fitted in models where relative year class strengths were assumed to be constant (i.e., equal to one) and, in these scenarios, tended to predict expected values that had a slower decline than the observed values (see Figure 20 for the CPUE fits for the female SSB model).

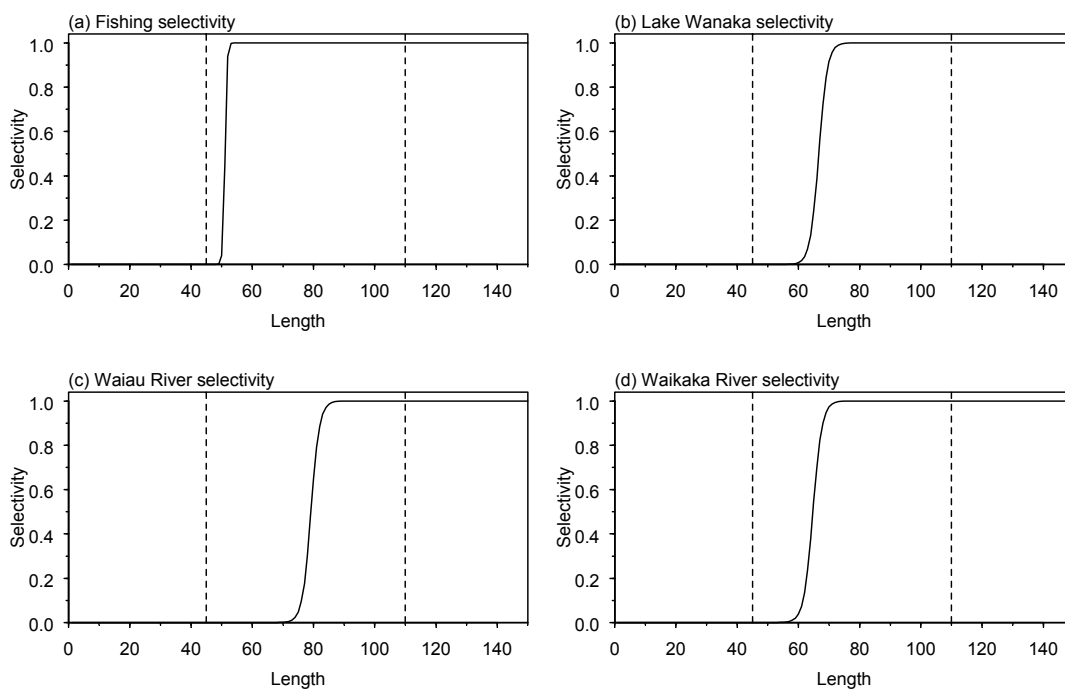
Growth estimates varied only slightly between model fits. Model fits to the age-length data are shown in Figure 21. Here, there was some suggestion from the data that growth rates may not be linear. And while model fits generally followed the pattern of increasing length at age, the variability around mean growth estimated by the models did not appear to capture the full range of variability within the data.

Exploitation rates estimated for all stocks were relatively high, with the values estimated for the female SSB model shown in Figure 22. In this case, the exploitation rates approached 0.3 in 2006, i.e., suggesting that about 30% of the vulnerable biomass has been taken as catch in the most recent year. A similar pattern of increasing exploitation rates in recent years was obtained for all of the models.

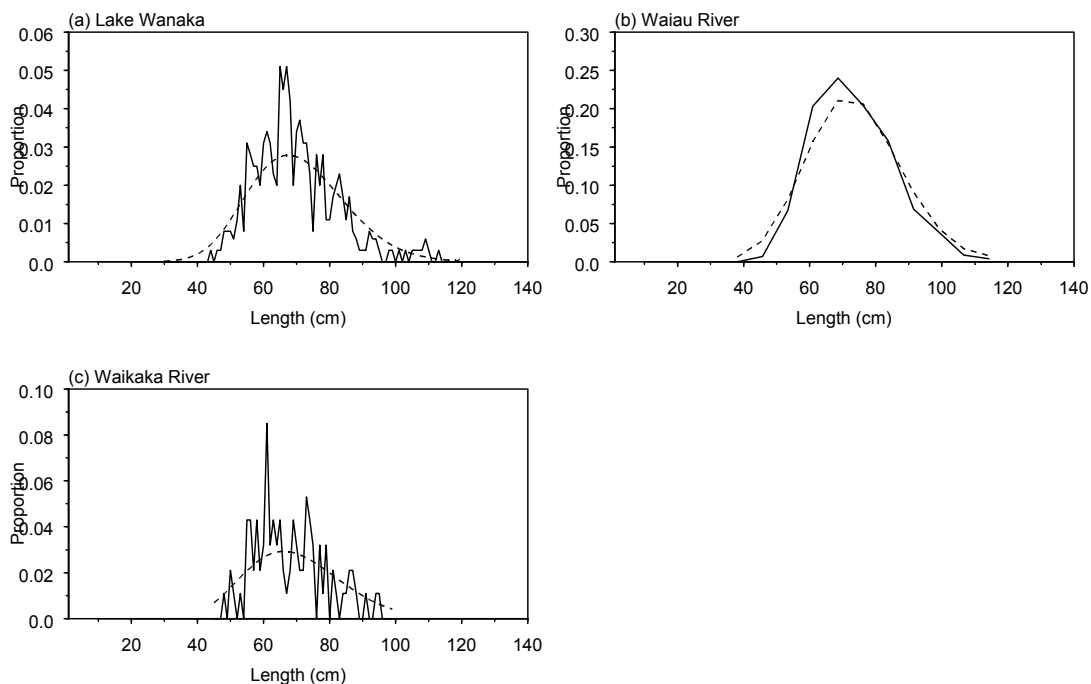
There were few differences in fits to scenarios with a single-area and those with two-areas, suggesting that there were insufficient data available to the models to determine which scenario was more likely.



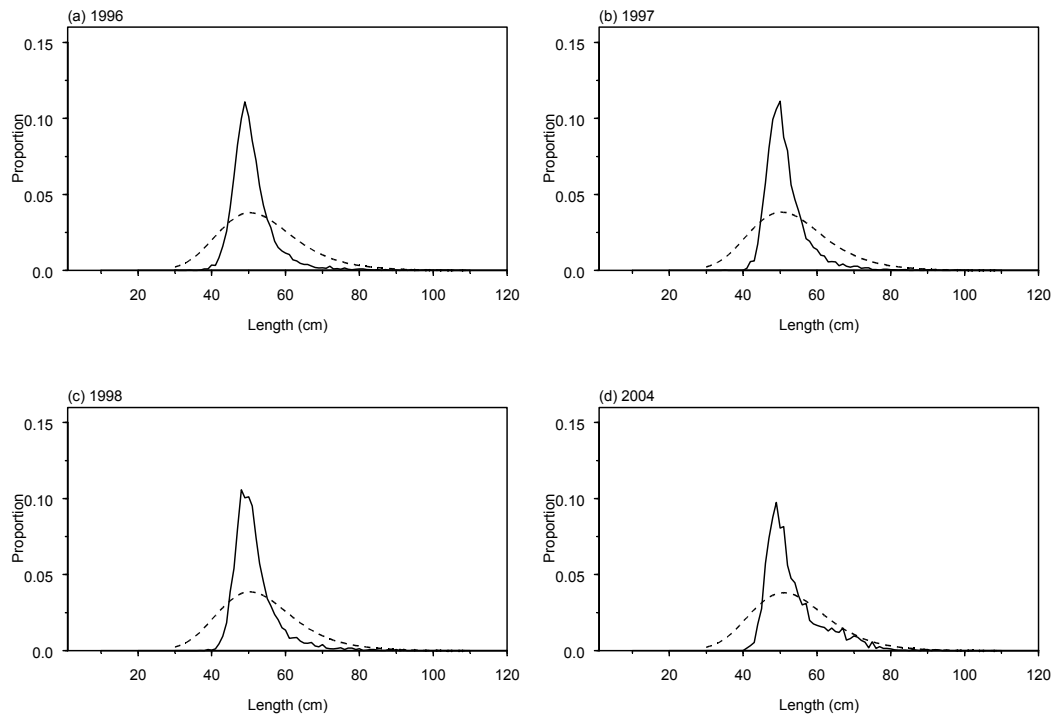
**Figure 14: Estimated MPD relative YCS for the YCS and the two-area YCS models.**



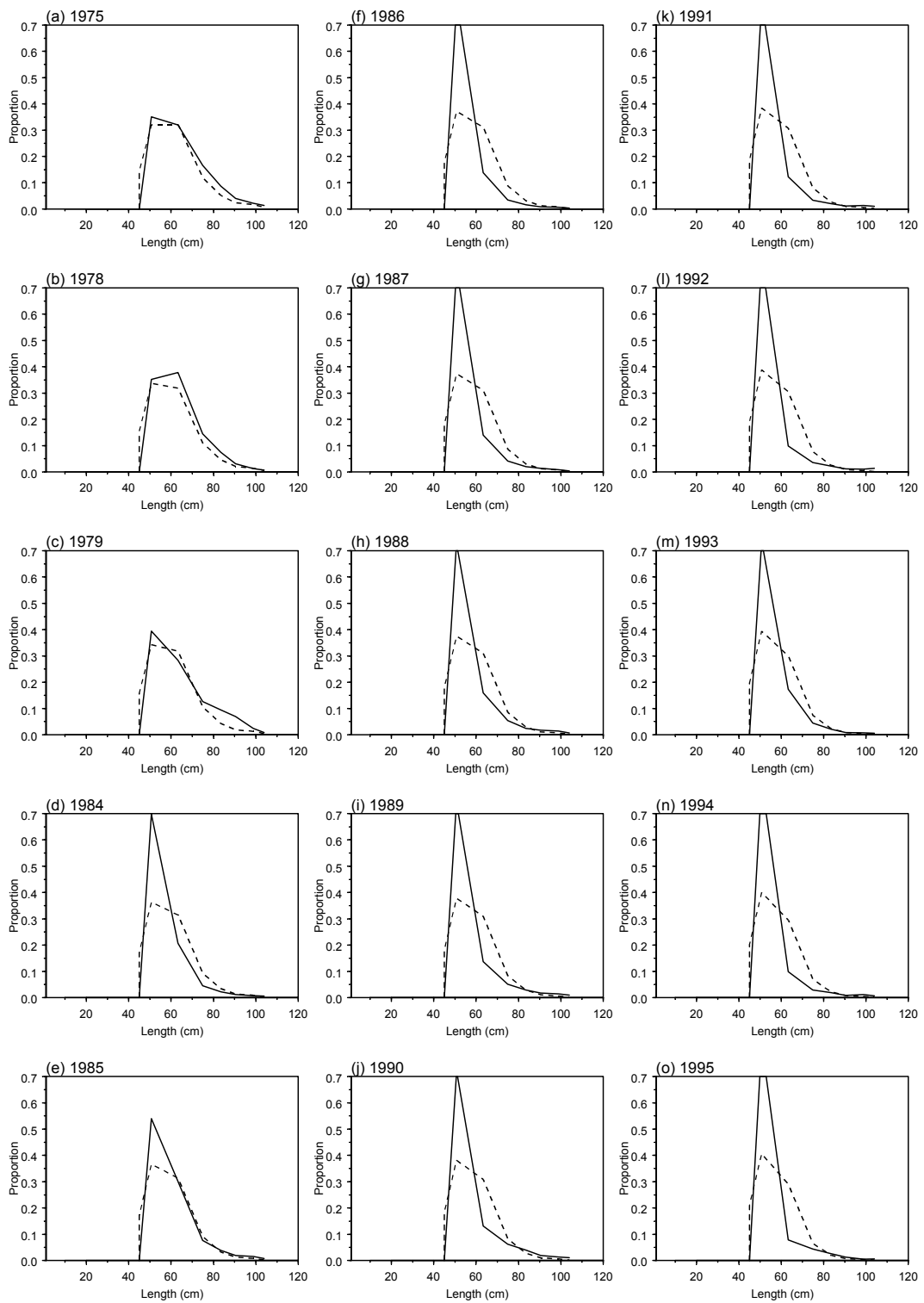
**Figure 15: Estimated MPD selectivities for (a) commercial fishery selectivity, (b) Lake Wanaka proportions-at-length, (c) Waiiau River proportions-at-length, and (d) Waikaka River proportions-at-length for the female SSB model.**



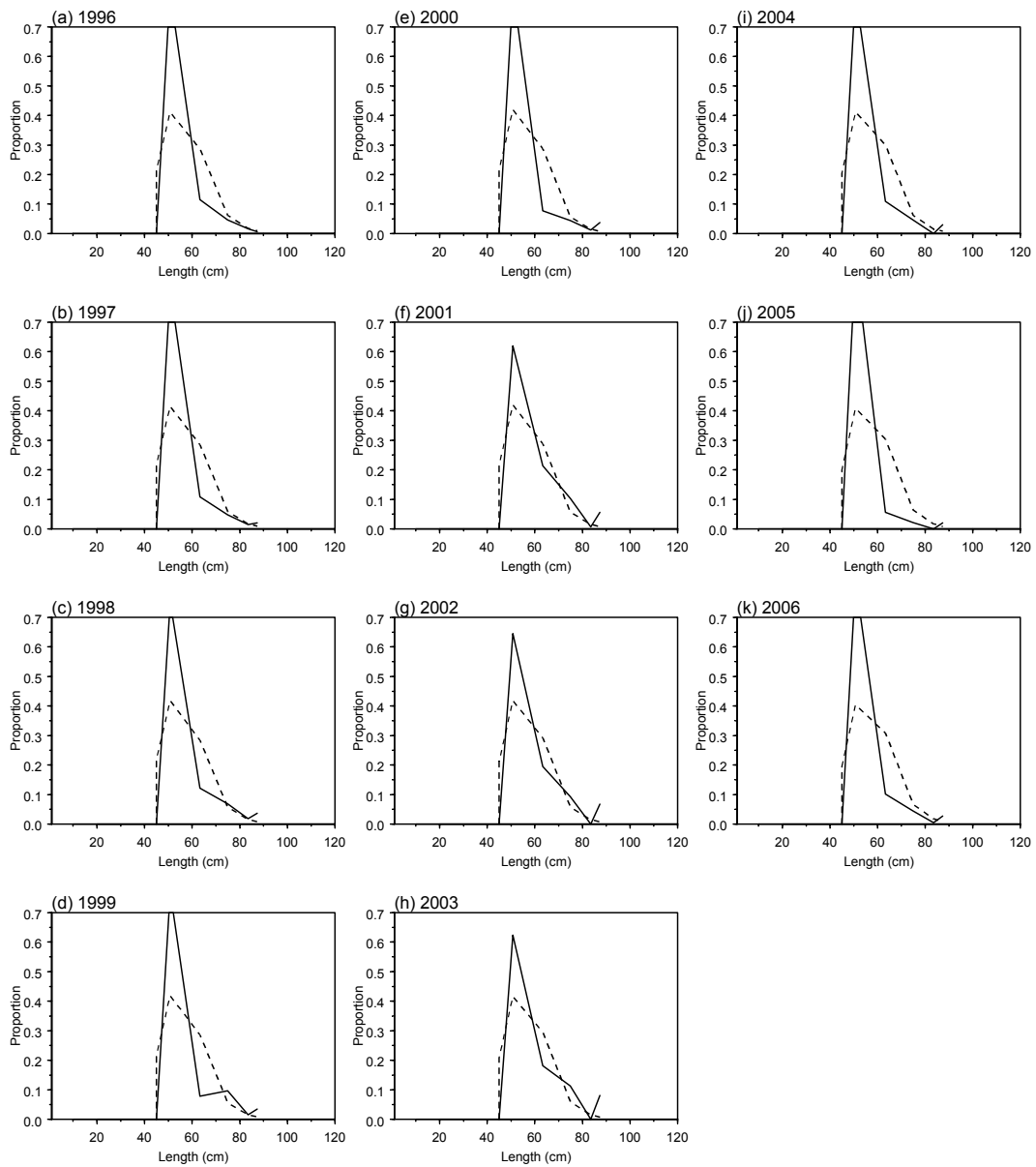
**Figure 16: Observed (solid lines) and expected (dashed lines) proportions-at-length for (a) Lake Wanaka proportions-at-length, (b) Waiiau River proportions-at-length, and (c) Waikaka River proportions-at-length for the female SSB model.**



**Figure 17: Observed (solid lines) and expected (dashed lines) proportions-at-length, 1996–2004, for commercial catch sampling length frequency data for the female SSB model.**

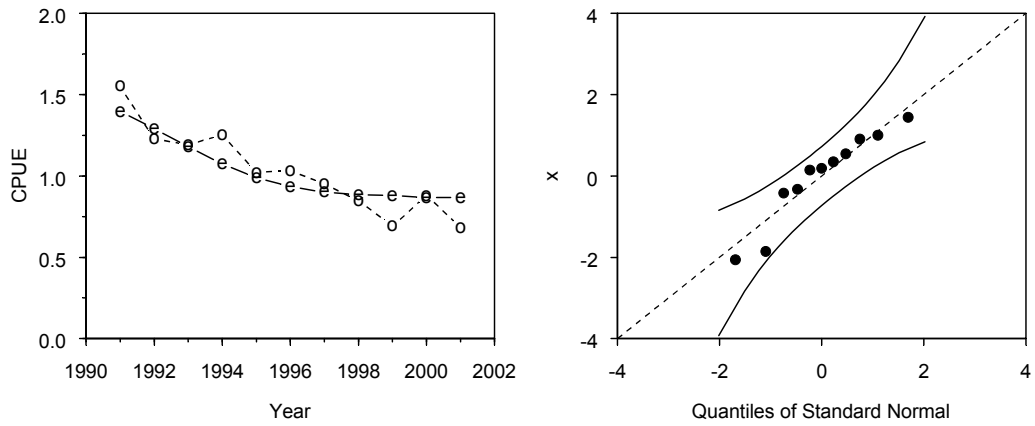


**Figure 18: Observed (solid lines) and expected (dashed lines) proportions-at-length, 1975–1995, for Mossburn Enterprises Ltd length frequency data for the female SSB model.**

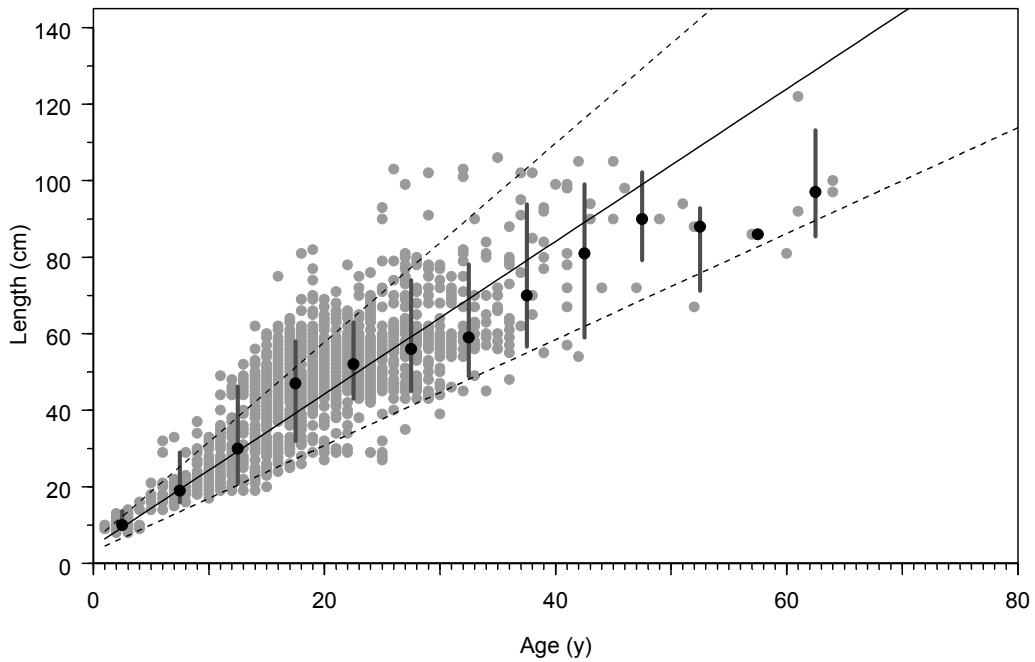


**Figure 19: Observed (solid lines) and expected (dashed lines) proportions-at-length, 1996–2006, for Mossburn Enterprises Ltd length frequency data for the female SSB model.**





**Figure 20:** Estimated MPD CPUE fits (left) and quantile-quantile plots of CPUE residuals (curved lines show approximate 95% confidence envelopes) for the female SSB model.



**Figure 21:** Estimated linear growth curve (solid line, with 95% intervals as dashed lines) for the age-length data (grey points) for the female SSB model. Point data medians for five year age classes are given as black points, with 95% ranges given as vertical lines.

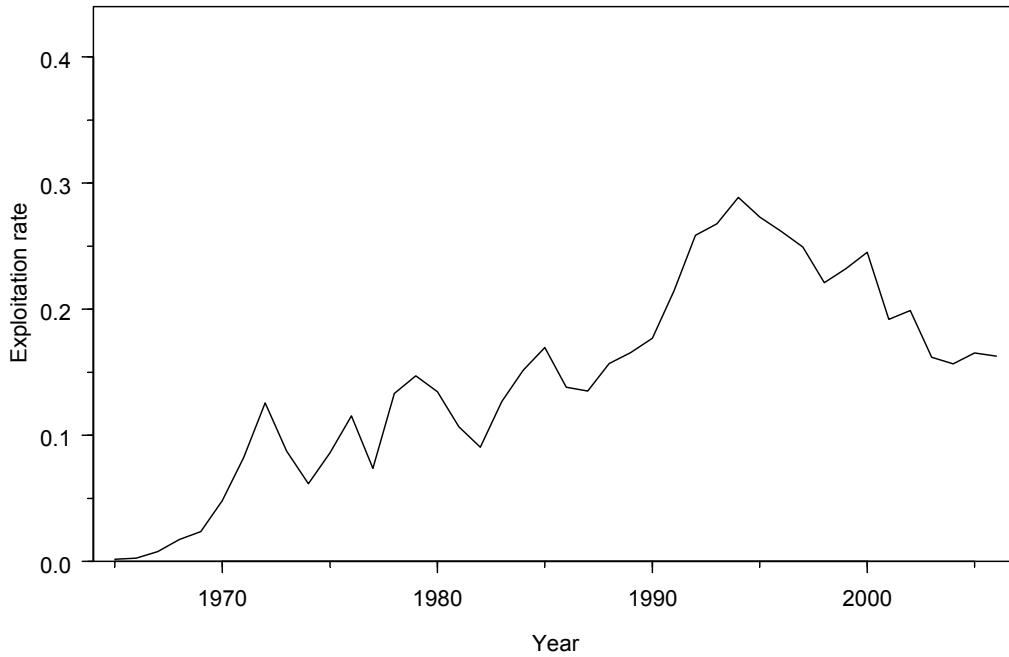


Figure 22: Estimated MPD exploitation rate for the female SSB model.

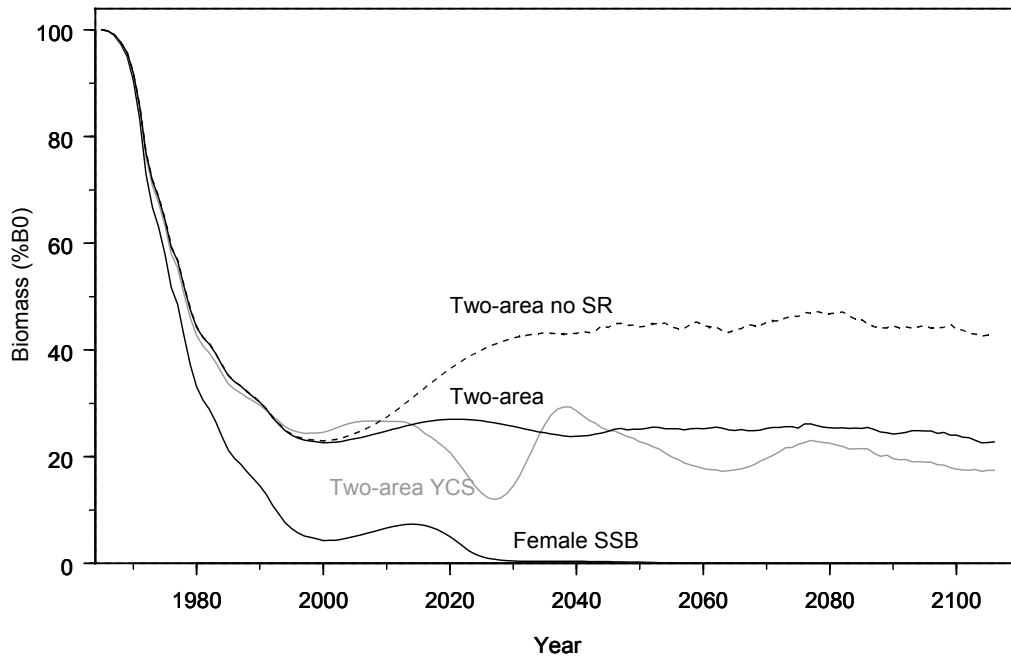
### 3.3 Projected stock status

Deterministic model projections were made for all models, assuming future recruitment variation with  $\sigma_R = 0.6$  for a 100 year period. In each case, 300 projections were made, randomly drawing recruits for 2001–2106 and projecting forward with a constant catch equal to that assumed for 2006 (i.e., a commercial fishery of 50 t, customary fishery of 12.5 t, and a recreational fishery of 1.25 t). The period of 100 years was chosen as recruits that enter the fishery in 2006 may take many years to grow large enough to migrate to spawn, and also to demonstrate how short-term populations may rise in response to pulses in recruitment or exploitation, but over the long term continue to decline.

Estimated spawning stock biomass for each model is given in Table 10, and the biomass trajectory for the female SSB and two-area models are shown in Figure 23.

Table 10: Selected median projected MPD estimates of  $B_0$ ,  $B_{2006}$ , and  $B_{2106}$  as a percentage of  $B_0$  and  $B_{2006}$  for all models.

Model	$B_0$	$B_{2006}$	$B_{2006} (\%B_0)$	$B_{2106}$	$B_{2106} (\%B_0)$	$B_{2106} (\%B_{2006})$
Initial case	182.2	39.0	21.4	37.8	20.8	97.0
Female SSB	122.4	6.5	5.3	0.0	0.0	0.0
YCS	119.8	6.6	5.3	0.0	0.0	0.0
CPUE c.v.	120.1	5.1	4.3	0.0	0.0	0.0
Two-area	149.5	35.2	23.6	34.0	22.8	96.4
Two-area, YCS	144.4	38.4	26.6	25.1	17.4	65.5
Two area, CPUE c.v.	147.8	34.1	23.1	31.4	21.3	92.2
Two-area, no SR	149.6	37.0	24.7	64.0	42.8	172.8



**Figure 23: Estimated median projected MPD SSB trajectories for selected model runs —female SSB, two-area model, two-area YCS model, and two-area no SR model.**

#### 4. DISCUSSION

We report the results of the first attempt at developing an age-structured population model for the longfin eel fishery. Estimates from the models in this report suggested that estimated pre-exploitation spawning stock biomass ranged between about 120 and 180 t, with the biomass of legal sized eels between 1700 t and 2100 t. Current spawning stock biomass was estimated to be about 5% assuming a single area, or 20–25% assuming two-areas were used. Simple projection suggested that, for the single stock models and at current catches, the SSB estimates from most models stayed at similar levels or declined slightly. The exception to this was when we assumed no stock-recruitment relationship — here, the stock underwent some rebuilding. In general, these models suggested that populations within fished regions of Southland are at depleted levels, and that the future spawning biomass of longfin eels stocks would appear to reside almost entirely in areas either closed or otherwise protected from fishing. If this is the case, then the levels of future recruitment of longfin eels may be entirely dependent on the relative size of these ‘reserves’ of stock biomass.

There have been a number of studies that have investigated the current state of the New Zealand longfin eel population. Hoyle & Jellyman (2002) reported that, for eel populations, with exploitation rates of 5% and 10% the spawning biomass would be reduced by 83% and 96.5% respectively. They concluded that, with current levels of exploitation and minimum legal size, longfin female eels were severely recruitment over-fished and that non-fishing reserves were required to offset the fishing pressure and to increase spawning escapement. Jellyman et al. (2000) reported that the cropping rate (= exploitation rate) for longfin eels may be as high as 20%, and at that level, few females would survive to spawn. They also concluded that longfin eels were recruitment over-fished. Graynoth et al. (2008b), using the Geographic Information System (GIS) method of Graynoth & Niven (2004), estimated biomass of eels in New Zealand catchments. They reported that, in areas open to fishing, the computer models and field studies indicated that relatively few female eels were left in fished areas and female escapement is derived mainly from reserves and unfished small streams. Doole (2005), using a bio-economic model applied to the longfin eel fishery in the Waikato River, concluded that longfin female eels were recruitment over-fished. Beentjes & Dunn (2003a) calculated CPUE indices for the Southland Fishstock, and, together with observations of changes in the length and sex ratios of eel populations from the main river fisheries (Beentjes 1999, Beentjes & Chisnall 1997, 1998,

Beentjes & Dunn 2003b), concluded that the changes in CPUE indices and sex ratios were consistent with a significant decline in the size of the populations within fished areas. Anecdotal reports have also suggested recent elver runs are less dense and frequent than were in the past. These papers have all reached similar conclusions, i.e., the longfin eel populations in fished areas are probably severely depleted. The results from this investigation, albeit preliminary, reached similar conclusions.

However, the model estimates presented here investigate only a limited range of model assumptions and hypotheses that may be plausible for longfin eel populations. We employed a generic integrated population modelling approach, based on those typically used to model commercially exploited marine finfish populations. While such an approach allowed an investigation of the key processes and parameters that may drive eel populations, we note that some population processes commonly thought to be important for eel populations cannot be easily accommodated within this model. Below, we attempt to identify those data, processes, and modelling assumptions that require further investigation, as well as outlining how such future work may be attempted. We also recall our earlier caution, and remind readers that the results presented here are predominantly for illustration, and may not be an accurate reflection of the current state of the Southland longfin eel population — some of the data used within these models show conflicting trends that have yet to be resolved, and further, these models are based on parameters and simplifying assumptions that have yet to be tested or validated.

Population models for eels are different from those usually applied to marine finfish species, but population characteristics such as slow growth, low natural mortality rates, and semelparous behaviour can easily be accommodated within suitably structured stock population models. Less straightforward is the inclusion of density dependent and local environmental effects (i.e., on recruitment, growth, sex determination, migration, and natural mortality). In addition, other sources of mortality or habitat change, such as pollution, loss of habitat, competition with introduced species, and obstructions to migration are difficult to model within the framework adopted here. While it has been assumed in the models presented here that these effects can be ignored, it is not known if inclusion of such processes within a model would result in different conclusions. We suggest that investigation of such simplifying assumptions would be useful. Here, simulation-based population operating models that include a range of density-dependent and local environmental effects could easily be developed, and the effect of such simplifying assumptions evaluated using estimation models that ignore such processes.

The choice of modelling Southland data alone is not ideal, particularly if the spawning stock biomass of longfin eels is made up from Fishstocks throughout New Zealand. However, these data allow us to investigate assumptions required to model New Zealand longfin eel populations, and allow discussion of the merits of an approach that may encompass all Fishstocks within a single-area model.

In general, we suggest that suitably structured models should be able to provide useful summaries of the current state of longfin eel populations, if the biological assumptions within the model are broadly correct, the data or observations reflect underlying population processes, the simplifying assumptions are valid, and if the models can be modified to recognise different areas with different exploitation histories.

Two spatial model structures were investigated in this report — single-area and two-area. The single-area models ignored that part of the population believed to reside in closed areas or areas that otherwise had lifelong protection from fishing. The two-area models assume a constant proportion of recruitment to a closed area. We note that within the CASAL model framework, the two-area approach can easily be extended to encompass a larger number of areas. Multiple area-based approaches could be used to reflect the lightly/heavily exploited areas and the protected area categories similar to those defined by Graynoth et al. (2008b), e.g., (i) waters that should not have been commercially fished (e.g. in National Parks) and have safe egress for migrating female eels; (ii) waters that are protected in their upper reaches but where migrants could be fished further downstream; (iii) waters that are located upstream of natural waterfalls or artificial dams that constrain the downstream or upstream migrations of either juvenile or adult eels; (iv) waters that are open to commercial fishing; and (v) streams that are

open to commercial fishing but are generally too small (less than 0.5 m<sup>3</sup>/s mean annual flow) to make fishing worthwhile.

Moreover, the use of area-based models may allow available data and observations for all New Zealand longfin eel Fishstocks to be integrated within a single-area model (assuming adequate observations in each that would allow model estimates to be made). We note, however, that for multiple-area models to be reliable, estimates or observations of the proportion of recruitment to each area (or alternatively estimates of the ratio of biomass in protected and fished areas) would be required.

The models presented here are age-based. However, there is some belief that many of the processes for eels are weight- or length-based. While an age-based model can approximate length based processes, this approximation may not be entirely satisfactory. We suggest that some thought be given to developing and comparing length-based models for longfin eels.

Population dynamics models, such as used in this report, require good estimates of the removals, or catch, taken from the population. While current estimates of commercial catch are believed to approximate the true level of catch from Southland, estimates of the historical customary and recreational catch, while likely to be small, are poor. The models used here assume that the customary and recreational catches are a constant proportion of the total commercial catch (25% and 3% respectively), but better estimates are required.

In addition, eel catches from areas above dams or structures that block or impede downstream spawning eel migrations (effectively taking eels from a part of the population that cannot contribute to the spawning stock biomass), upstream elver migration, or the effects of other non-fishing mortality may not have been adequately accounted for in these models. Estimates of the proportion of catch from different types of areas, and how this proportion has changed over time, may improve the accuracy of these models.

While estimates of rates of natural mortality for longfin eels are available (e.g., Jellyman 1995), it is not known how good these estimates are. In addition, the estimates assume a constant rate of natural mortality with age/length and over time. Some evidence exists for density dependent and environment dependent rates of natural mortality in European eels (De Leo & Gatto 1995, Vøllestad & Jonsson 1988). In a recent study, Graynoth et al. (2008a) investigated factors that influenced survival of juvenile longfin eels (i.e., eels under 40 cm) in three lowland New Zealand streams. They reported that the proportion of 0+ glass eels that survived to 40 cm in length was highly variable and that the decline in survival rates for large juvenile eels supported a habitat bottleneck hypothesis, possibly caused by competition from adult eels for limited areas of suitable cover or food. We note that estimates of the nature and type of variability in rates of natural mortality, as well as the assessment of the effect of density-dependent natural mortality on assessment models will need to be investigated in future models.

The maturation/migration rates used here were based on estimates from Todd (1980), Beentjes & Chisnall (1998), and Beentjes (1999, 2005), and, as noted by Todd (1980), appear to vary between regions across New Zealand. Further investigation of the rates of maturation/migration, as a function of length, weight, or age, needs to be undertaken. In addition, source data observations of the proportions-at-length migrating may be better included as observations, with maturation/migration rates estimated within the models. We also note that the two-area model structure proposed here could plausibly be altered to allow for different rates of maturation/migration by length or age between areas.

Little is known of the nature of any eel species stock-recruit relationship. A depensatory relationship has been proposed for the European eel (International Council for the Exploration of the Sea 2005). If the stock-recruitment relationship is depensatory, then it is plausible that New Zealand longfin eel stocks may already be below the level that would allow for the current stock size to be maintained. We

have assumed a Beverton-Holt stock-recruit relationship within these models, but note that such relationships are highly uncertain and the evidence that exists for, or against, any stock-recruit relationship is weak. We note that alternative stock-recruitment relationships are plausible and should be investigated in future work. Further, the relative importance of using either total, male only, or female only biomass will need to be considered.

We assumed that the ratio of male to female eels was 50:50 at the time of recruitment to the model (i.e., age 1). We note that the 50:50 ratio assumption is arbitrary, and also that some evidence exists for density-dependent sex determination in New Zealand eel populations (Beentjes et al. 2006, Davey & Jellyman 2005). Future work should include investigation of the performance of single sex models (although such an approach would ignore the sex structure of spawning biomass), or the inclusion of sex ratio observations that may allow the ratio of male to female recruitment to be better estimated. We also note that the generic model CASAL (Bull et al. 2005) cannot easily be modified to address density-dependent sex determination processes.

The proportions-at-length observations used in these models appear to be in conflict. The commercial catch sampling data (Beentjes 1999, 2005, Beentjes & Chisnall 1997, 1998) suggest very few eels above about 80 cm in the commercial catch, while the proportions-at-weight data from Mossburn Enterprises Ltd suggest that a significant proportion of eels above 80 cm were present. There are at least two plausible explanations for the apparent conflict — the first that the commercial catch sampling data are biased, and/or second, that the method used to convert the proportions-at-weight data was inadequate. We have noted that conversion may not be ideal (e.g., it did not take into account changes within weight classes that may have resulted from exploitation), but we were unable to include proportions-at-weight observations without modifications to the modelling software. (Although such changes are relatively straightforward, they can be time consuming to implement and validate). However, given that the catch sampling data were based on length-frequencies derived, not from a sample of the total catch, but from a sample of catch taken from heavily fished areas, there is some evidence for bias in the commercial length frequency data.

There is considerable evidence for changes in the sex ratios of longfin eels in fished areas over time (Beentjes et al. 2006, Davey & Jellyman 2005, McCleave & Jellyman 2004). Reports from early sampling suggested that there was a predominance of larger female longfin eels in the 1940s and 1950s. However, more recent catch-sampling has suggested a predominance of male longfin eels. Sex ratio data were not included within these models, as pre-mature eels are often of indeterminate sex, and observations that consist of a combination of unsexed, male, and female proportions cannot be included within the currently available modelling software. However, we recommend that methods that would allow such data to be included within these models be investigated in future developments.

Fishing selectivities for the commercial, customary, and recreational catch were all assumed to be the same and ‘flat topped’ (i.e., no declining right hand limb). The introduction in 1996, of a maximum weight limit of 4000 g might suggest that a right-hand limb declining selectivity may be more appropriate, at least for catches after 1995. However, if the stock is as depleted as these models would suggest, the inclusion of domed selectivity ogives would be unlikely to have altered the outcomes of these model runs. We also note that the 2006 Plenary Report (Ministry of Fisheries Science Group 2006) states that ‘Customary fishers desire eels of a greater size, over 75 cm and 1 kg. These size/weight preferences may imply a different selectivity than that for the commercial fishery.

The proportions-at-length data for Lake Wanaka, Waiiau River, and Waikaka River provide useful information on the expected length frequency of unexploited eels. The fits to these data were, in general, reasonable. We would recommend that similar data sets be collated for other longfin eel fishstocks.

The sample size estimates (or likelihood weightings) used for the CPUE and proportions-at-length data used for the models were arbitrary, although chosen so as to give roughly equal influence to each type of data set. Likelihood weightings for each of the proportions-at-length data sets were set at

$n = 50$ , and CPUE process errors at 10%. Future work will need to investigate approaches to determining relative weightings, and hence determine the levels of data weights that should be applied to each of the sets of data.

The CPUE indices provide the only real abundance data available to these models, given the historical catches and the decline suggested by the CPUE indices up to 2001, model outcomes that suggest, at least in fished areas, that the populations within fished areas are not in a highly depleted state are unlikely. However, the quality of the CPUE data for individual species (longfin and shortfin) is questionable in some areas, because the code 'EEU' (i.e., unidentified eel species), rather than species specific codes, was often used before 2001. However, in ESA20/AW, the use of the EEU code was less significant than in other areas, and here the data were believed to be of a reasonably high standard. We note that an update of the CPUE indices for the South Island (including Southland) is in progress, and may be useful to include in future model developments.

However, it is not known if CPUE indices reflect changes in abundance for eels. In marine finfish populations, there is often a belief that CPUE indices are hyper-stable, i.e., the underlying population abundance has declined at a rate faster than suggested by the CPUE indices. The reasons for such a belief are that changes in gear, technological improvements (including the introduction of Global Positioning Systems and side-scan sonar), and other changes in fishing practice allow fishers to maintain catch rates as fish abundance declines. Such changes are less likely to be an influence in freshwater fisheries such as eels.

There are few data on the recruitment of glass eels and elvers into New Zealand freshwaters, although the 2006 Plenary Report (Ministry of Fisheries Science Group 2006) reports that glass eel runs were estimated to be a quarter of the size of runs before the early 1970s. Data on relative elver counts from the mid 1990s were used in this model as relative indices of recruitment for longfin eels. However, these time series are relatively short when compared with the time series of catch history. In these models, the elver estimates were converted to an annual index and included within the model as observations of relative indices of numbers of eels with a selectivity assumed to be 1.0 for eels aged 1, 2, and 3, and zero otherwise. However, the actual selectivity of the elver data is not well known, and observations on either the age or length frequency of the elver data should be collected. In general model fits to these data were poor, and the best method for including such data may need to be investigated.

In general, some of the age data available to these models did not appear well fitted with a linear growth model. The data appeared to show some non-linearity, particularly at younger ages/lengths. The reasons for poor fits are not known, but may be confounded by inconsistent ageing, poor assumptions as to the nature of the ageing error, poor or biased sampling data, density or environmentally dependent growth, or simply that the growth curves may not be linear over the range of ages modelled here. Better age data are required for eels, particularly over time and between catchments in order to better understand longfin eel growth curves. We also note that the age-length data from Southland catch sampling length-at-age (Beentjes 1999, 2005, Beentjes & Chisnall 1998) and that from the Aparima River catchment (Graynoth 1999, McCleave & Jellyman 2004) appear to be in conflict, and may need to be investigated in future models.

Even so, the growth estimates from the modelled age data compare reasonably with those proposed by Horn (1996). The female SSB model growth estimates suggested that a longfin eel reached minimum legal size (45 cm) at a mean age of 20 years (and 95% range 15–30 years). Beentjes & Chisnall (1998) proposed that longfin eels reached the minimum legal weight of 220 g (roughly equivalent to 45 cm) on average, at 17.5 years (range 12.2–28.7 years). Todd (1980) reported the range of recorded age and length at migration was 25–60 years and 74–156 cm for females (age data for males were not reported). From the growth model for the female SSB case, the upper 95% range for eels of age 25 was 74 cm, and for eels of age 60 it was 161 cm.

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