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Monitoring of eel populations using mean size

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This series documents the scientific basis for stock assessments and fisheries management advice in New Zealand. It addresses the issues of the day in the current legislative context and the time frames required. The documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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1 Executive Summary

This document is a final report on work carried out under Objective 5 of Ministry of Fisheries project EEL9701: *To assess the feasibility of determining the current status of eel stocks in the priority eel fisheries by analysis of size frequency data.*

Information on the dynamics of New Zealand and other freshwater eel populations was used to construct a simulation model of an eel fishery. Various exploitation scenarios were simulated to see whether, and how easily, a change in stock status from “biomass stable” to “biomass declining” could be detected using only size data.

It was found that size data, by themselves, could be useful for detecting large long-term changes in stock status, but are not likely to be good indicators for use in year-to-year management of eel populations. Natural variations in glass eel recruitment cause substantial year-to-year variations in the mean size of eels, even when fishing pressure is constant. This makes it difficult to detect changes in mean eel size caused by long-term changes in fishing pressure. Even without sampling error, biomass would need to drop by more than 40% (or more, if the drop is gradual) before this change could be reliably detected. The presence of typical levels of sampling error makes this detection even harder.

This conclusion cannot be considered definitive because there is little detailed information about New Zealand eel population dynamics. A number of phenomena (such as annual variation in growth and mortality, and density-dependent growth) which are known to occur in eel populations were not included in the population model for lack of data. However, there is reason to believe that the inclusion of at least some of these would make the above result more, rather than less, pessimistic.

No evidence could be found of the use of size data to monitor eel stock status in other parts of the world.

It remains to be established how much more effectively New Zealand eel populations could be monitored with the inclusion of catch-at-age data.

2 Introduction

The commercial fishery for freshwater eels (*Anguilla australis*, *A. dieffenbachii*) in New Zealand has been operating since the 1960s. However, until very recently there has been only very limited information to support fishery management. This lack of information was one reason for initiating, in 1995–96, a sampling scheme in processing sheds aimed at obtaining information (size, age, and species composition) about the catch from the main fisheries. The scheme has provided much-needed baseline information for these fisheries (Beentjes & Chisnall 1997, 1998).

This document considers another possible use of data from this scheme: to monitor eel fisheries for management purposes. It addresses Objective 5 of the Ministry of Fisheries project EEL9701: *To assess the feasibility of determining the current status of eel stocks in the priority eel fisheries by analysis of size frequency data.* Thus, its aim is to determine whether size data gathered in a shed-sampling scheme could be useful in monitoring eel fisheries. This fits in to the overall (programme) objective for EEL9701: *To assess and monitor commercial eel fisheries.*

It is not unreasonable to assume that size data alone could be a useful indicator of stock status. As the biomass of a fish population is reduced by fishing it is to be expected that the mean age will also decrease (with higher fishing pressure, fewer fish survive to old age). Because age is highly correlated with size (length or weight) it might also be expected that mean size would decrease with decreasing biomass. This relationship is the basis of methods for estimating total mortality from mean length (Beverton & Holt 1956, Wetherall *et al.* 1987).

However, there are two factors that may prevent shed-sampling size data being a useful indicator of stock status. First, even when there is no trend in biomass, size distributions may be expected to fluctuate from year to year (because, for example, of random variations in recruitment). If these fluctuations are of sufficient magnitude they will make it difficult to detect trends in biomass. Second, the sampling may not be sufficiently intense to reliably detect the changes in size distribution associated with substantial changes in biomass. In this report we use a simulation approach to investigate the effect of these factors and thus determine how useful size data may be as an indicator of stock status.

The first step was to assemble information about biological parameters of eel populations. This information was then used to construct a stochastic model of an eel fishery. Then simulations with this model were used to address the relationship between eel size and stock status. Finally, the international literature on eel management was reviewed to seek examples of eel fisheries monitored using size data.

3 Biological parameters

For the purposes of the simulations below it was necessary to determine: a) the most appropriate way to describe mathematically various biological processes (e.g., growth, maturity, and recruitment) in New Zealand freshwater eel populations; b) typical values for the associated biological parameters; c) the degree to which these parameter values might vary from stock to stock; and d) how much the parameter values might vary from year to year for the same stock. Where this information was not available for New Zealand eels, data from other freshwater eel species were used.

Four sources of data were used.

1. Data from the 1995–96 and 1996–97 shed-sampling programmes;
2. NIWA's Freshwater Eel Database of almost 40 000 eels from various sampling programmes from 1974 to the present

3. A literature review of published information for New Zealand eels (see Appendix); and
4. Selected published material on other freshwater eel species.

3.1 Growth

There is a large number of published estimates of growth (mean length as a function of age) for New Zealand eels (see Appendix 1). In almost all cases, a linear growth curve has been used, and no distinction has been made between the growth of males and females (macroscopic identification of the sex of immature eels is difficult). For present purposes it was important to know not only the mean length at each age, but also how much the lengths of individuals of the same age may vary. For this reason all the following inferences about growth derive from data in the NIWA database. An examination of these data suggested the following model, in which the length, L , of an eel of age i is given by

$$L = a_1 + b_1 i + (a_2 + b_2 i)\varepsilon \quad (1)$$

where the parameters (a_1, b_1) define mean length as a linear function of age, (a_2, b_2) describe the standard deviation of length-at-age, and ε is a standard normal variate representing between-individual variation in length at age. (Throughout this report we follow the usual convention for eels that “age” always refers to age in fresh water, so that glass eels have age 0).

With this model, the mean and standard deviation of the lengths of eels of age 0 are a_1 and a_2 , respectively. Because data to estimate these parameters for New Zealand eels are very scattered (across both time and locations), and because they appear very similar for the two species, it was decided to pool all the data. The resulting estimates of a_1 and a_2 were 8.1 cm and 1.2 cm, respectively (with estimates for the individual species being 8.3 cm and 1.0 cm for longfins, and 8.0 cm and 1.3 cm for shortfins).

The growth model (1) was fitted to data sets from all combinations of location and species with at least 100 age-length pairs (Figure 1). The parameters a_1 and a_2 were fixed at 8.1 cm and 1.2 cm, respectively, and b_1 and b_2 were estimated by maximum likelihood. The effect of outliers was minimised by making an initial fit, removing all points which deviated from this fit by more than 3 standard deviations, and then refitting the model. The extent to which model parameters (and thus growth) may vary between locations and species is shown in Figure 2. In general there is more variation amongst locations than between the two species. There is a tendency in this plot for the points associated with shortfins to be above the regression line and the longfin points to be below (which would imply that, for a given mean growth rate, b_1 , length at age is more variable for shortfin than for longfin). However, this effect could be an artefact of the particular locations for which samples were available for each species.

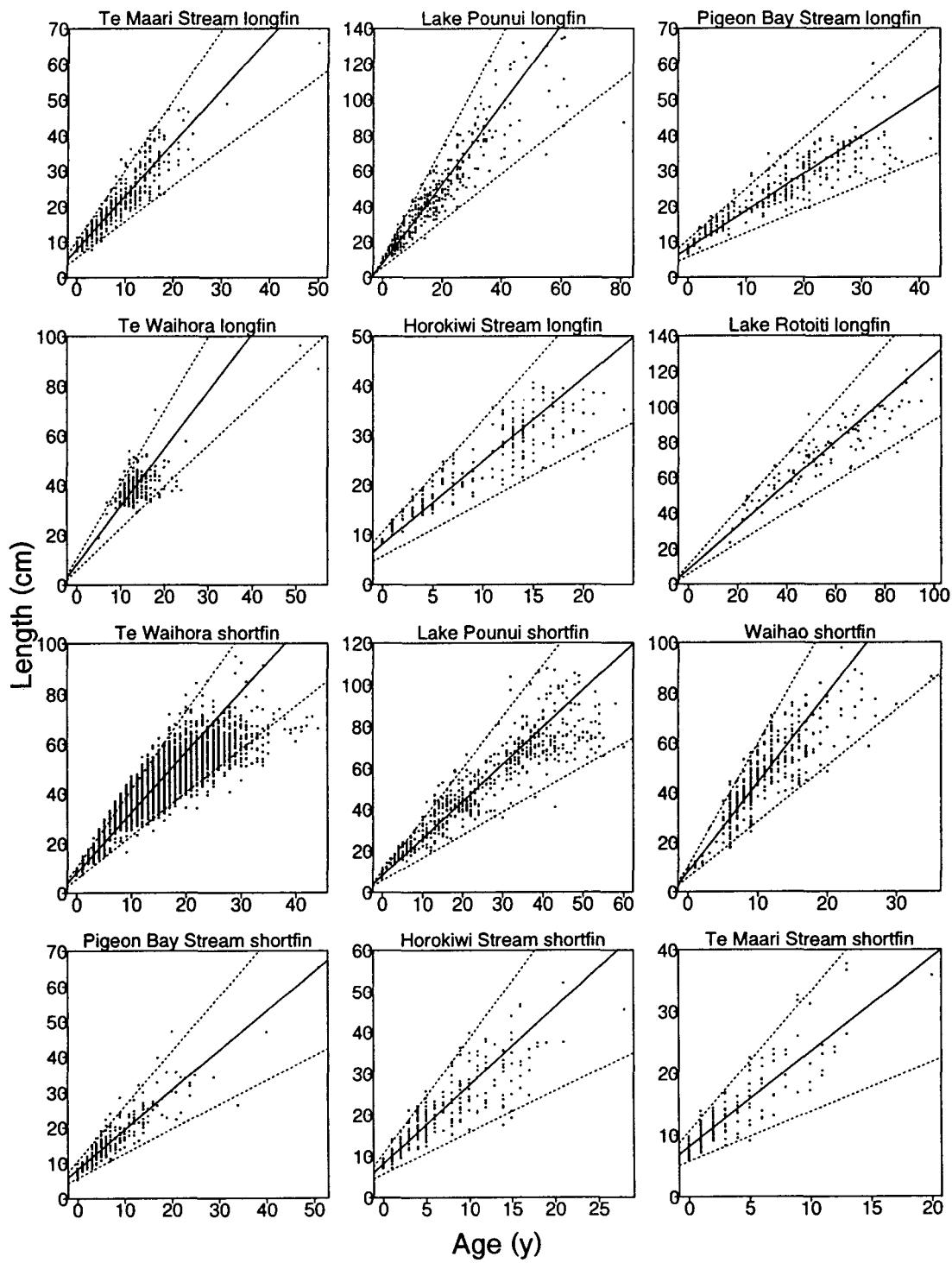


Figure 1: Fits of the growth model (1) to data sets from all combinations of location and species with at least 100 age-length pairs. Solid lines show mean length at age; broken lines show mean \pm 2 standard deviations.

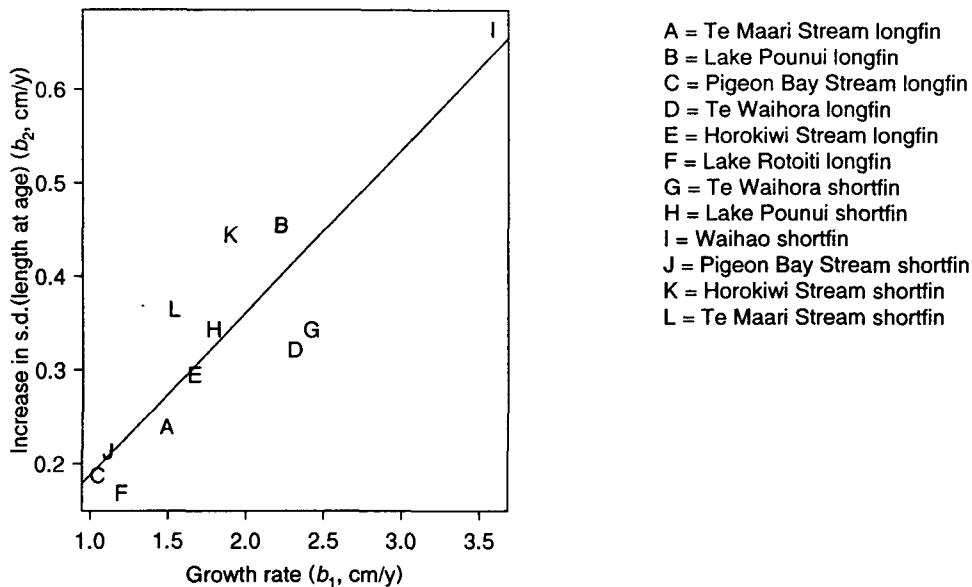


Figure 2: Plot of growth parameters b_2 vs b_1 for the fits to the 12 data sets in Figure 1. The regression line is $Y = 0.01 + 0.17X$.

3.2 Length-weight

The mean weight on an individual of length L is conventionally modelled (for eels as well as other fish species) as being given by αL^β . The shed-sampling data fitted this model well, and estimated parameter values were $\alpha = 1.18 \times 10^{-3}$, $\beta = 3.18$ for longfins and $\alpha = 1.48 \times 10^{-3}$, $\beta = 3.08$ for shortfins (where length is in centimetres and weight in grams). (This means that longfins are typically heavier than shortfins of the same length, with the difference rising from 16% at 40 cm to 27% at 100 cm).

Both the shed-sampling and NIWA databases were examined for evidence of year-to-year variation in this relationship at a given location. This variation was found to be small – typically less than 1%, and always less than 5%.

3.3 Recruitment

It is believed that recruitment of New Zealand eels is highly variable between seasons and between areas (Annala *et al.* 1998) but existing time series are too short to extract reliable statistics (see Appendix 1). However, we were able to assemble series of annual recruitment indices for a number of other eel populations (Figure 3). These series ranged in length from 10 y to 52 y (median 19 y) and have mainly come from commercial glass eel catch rates (stocks 1–18) or elver passes on hydroelectric dams (stock 20).

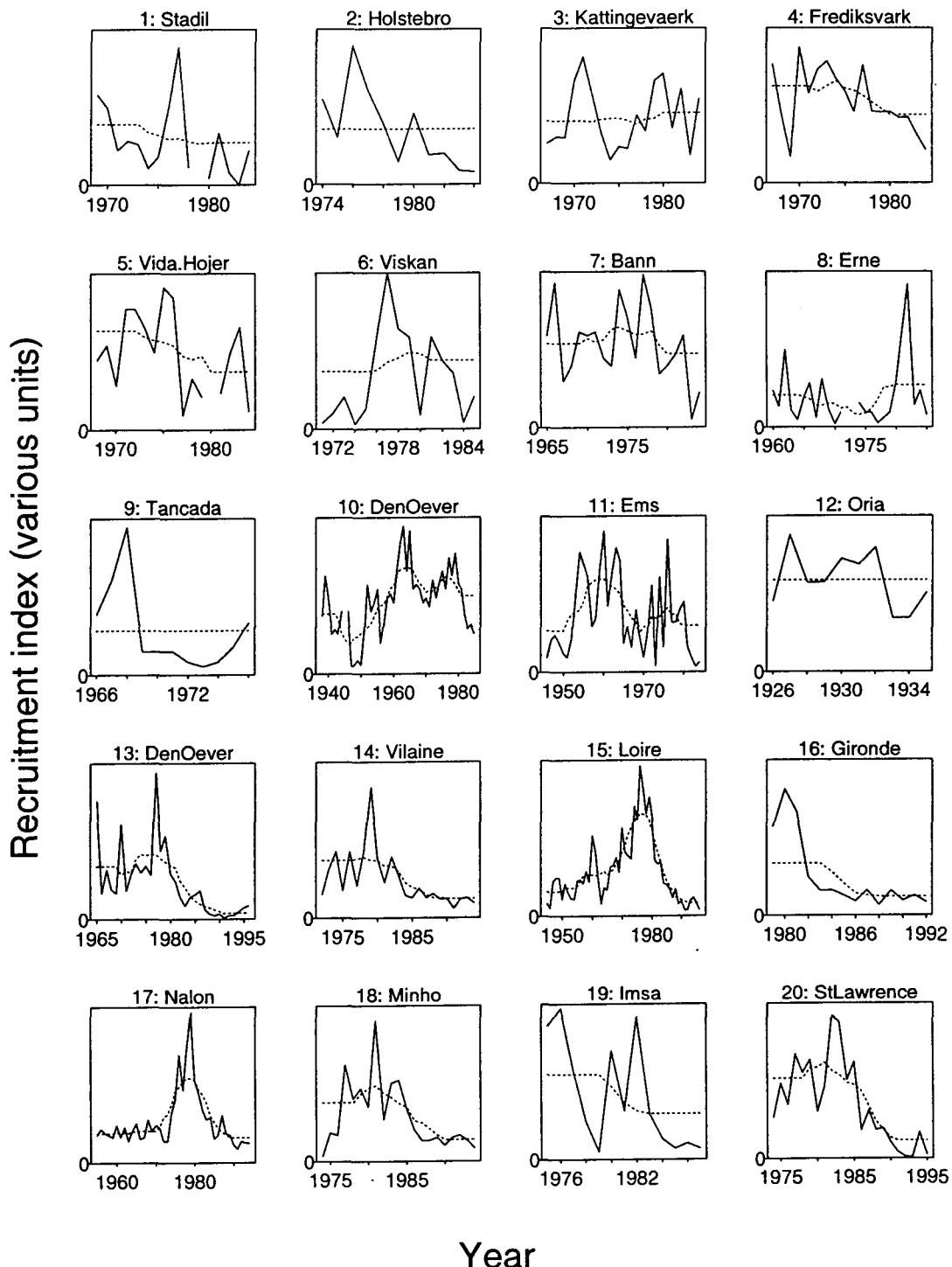


Figure 3: Sequences of recruitment indices for 20 stocks of eel. The broken line is a 9-year moving average.
 Sources: stocks 1–11, Moriarty (1986); stocks 12–18, Desauny & Guerault (1997); stock 19, Vøllestad & Jonsson (1988); stock 20, Casselman *et al.* (1997). All are the European eel, *Anguilla anguilla*, except for stock 20, which is the American eel, *A. rostrata*.

We chose to model log recruitment as an AR(1) (first-order autoregressive) process (Box & Jenkins 1976). The reasons for this choice are: fish recruitment is often treated as independent lognormal but there is clearly autocorrelation in the series in Figure 3 and this autocorrelation is likely to affect year-to-year changes in mean eel size; the variable quality and length of these series do not seem to justify the estimation of more than two parameters for each series. Before fitting this model (by maximum likelihood) to the data of Figure 3, each series was detrended (by dividing by a 9-year moving average) and then logarithms were taken. (The length of the moving average was somewhat arbitrary. A longer value would risk leaving some residual trend, and thus over-estimating variability and auto-correlation; a shorter value could lead to under-estimating auto-correlation.) The estimated AR(1) parameters for each stock (the standard deviation, σ_R , and lag-1 autocorrelation, ρ_R) are plotted in Figure 4. These covered a wide range of values: from 0.3 to 1.0 (median 0.6) for σ_R , and from -0.1 to 0.6 (median 0.2) for ρ_R .

There are some grounds to believe that eel recruitment variability in New Zealand is at the upper end of the range covered by the data sets in Figure 4. In both the New Zealand recruitment series (Appendix Tables 1 & 2) the maximum ratio of recruitments in successive years is about 9 (both series have length 4). To see how likely this would be with the degree of variability shown in Figure 4, 1000 4-year AR(1) sequences of recruitment were generated for each of the 20 (σ_R , ρ_R) pairs of parameters. The proportion of sequences where the maximum ratio in successive recruitments exceeded 9 was zero for seven pairs, and was never greater than 0.15.

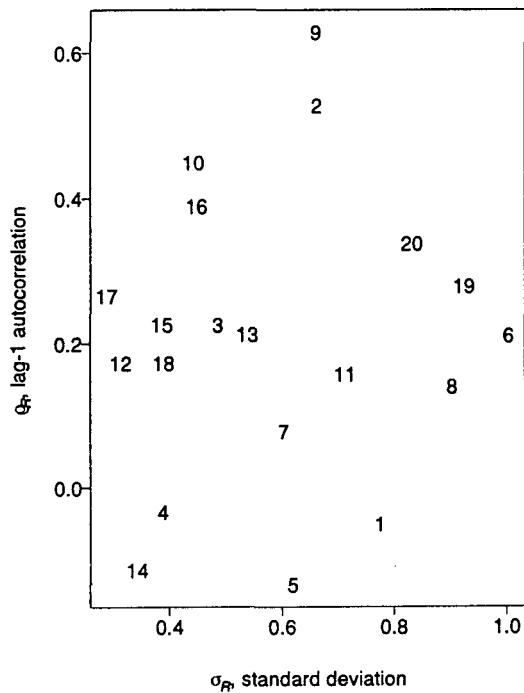


Figure 4: Recruitment statistics, σ_R and ρ_R , for twenty eel stocks (see text for details of calculation). Plotting symbols refer to the stock numbers in Figure 3.

3.4 Natural mortality

There are only three estimates of natural mortality, M , for New Zealand eels: 0.042 for longfins in Lake Rotoiti (Jellyman 1995), and 0.038 and 0.036 for shortfins and longfins, respectively, in Lake Pounui (authors' unpublished data). These estimates are all for 'large' eels (longer than about 40 cm).

3.5 Maturity

Todd (1980) gave statistics on the length and age at maturity (or migration to the sea), by species and sex, at three New Zealand locations. However, it is only for shortfins at Te Waihora (Lake Ellesmere) that sample sizes for both males and females were substantial. Here (and for all other species-location combinations) he found the common pattern amongst migrating eels that females were larger and older than males (Table 1).

Table 1: Statistics on length and age (by sex) at maturity for shortfin eels at Te Waihora (from Todd 1980; further statistics from this source are given in Appendix Table 4).

	Length (cm)		Age (y)	
	M	F	M	F
Mean	43.2	60.9	14.4	23.6
Standard deviation	2.2	9.3	2.0	3.6
Sample size	12 020	778	1263	181

De Leo & Gatto (1995) modelled the maturity of eels as being a probabilistic function of length. They assumed that the probability that an eel of length L will become mature that year is given by

$$\frac{\gamma}{1 + \exp\left(\frac{\lambda - L}{\eta}\right)} \quad (2)$$

Where the parameters γ , λ , and η are different for the two sexes. This model has the merit of being consistent with the observations that: eels that migrate at a younger age than average are typically faster growing than those that don't; and, on average, the older migrants are, the bigger they are (Todd 1980, Vøllestad & Jonsson 1988).

It is not straightforward to infer parameter values for model (2) from statistics like those in Table 1. This is because these statistics depend not only on these parameters, but also on growth and mortality. We used the following procedure to estimate γ , λ , and η for shortfin eels in Te Waihora. Given trial values of γ , λ , and η , together with the estimated growth parameters for this stock from above ($a_1 = 8.1$ cm, $a_2 = 1.2$ cm, $b_1 = 2.42$ cm y^{-1} , and $b_2 = 0.34$ cm y^{-1}) and an assumed total mortality of 0.04, we used a straightforward simulation to calculate the mean and standard deviation of the length and age at maturity, $L_{m\bar{v}}$, $L_{s\bar{d}}$, $A_{m\bar{v}}$, $A_{s\bar{d}}$. We then calculated the difference between these "estimated" statistics, and the "observed" statistics in Table 1 as

$$\sum_i \left| \frac{O_i - E_i}{O_i} \right|$$

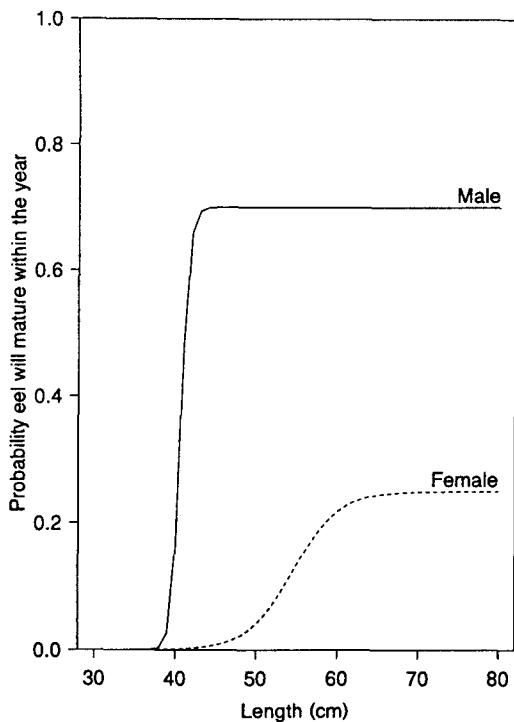
where O_i and E_i are the i th observed and expected statistics, respectively. Then a

search procedure was used to find the values of γ , λ , and η that minimised the above difference. As expected, the estimated parameters are quite different for males and females (Table 2), as are the associated maturity curves (Figure 5).

Table 2: Estimated values of the maturity parameters (for equation (2)) for shortfin eels in Te Waihora

Parameter	Value	
	Male	Female
γ (no units)	0.7	0.25
λ (cm)	40.6	54.6
η (cm)	0.5	2.8

Figure 5: Estimated maturity curves for male and female shortfin eels from Te Waihora. Each curve shows the estimated probability that an eel that reaches a given length at the beginning of a year will mature and migrate within that year.



4 The fishery model

A model of an eel fishery was constructed that was based, as much as possible, on the conclusions drawn from Section 3. It has stochastic recruitment, individual variation in growth and initial length, and models maturity using the approach of De Leo & Gatto (1995) (i.e., equation (2) above).

The number of recruits (glass eels) each year was generated using an AR(1) process with parameters σ_R and ρ_R , and the sex-ratio of glass eels was assumed always to be 50:50.

Growth was modelled using a random-coefficients model (i.e., all eels are assumed to follow the same growth curve, but each individual has its own growth parameters, which do not vary during its lifetime). However, we used the linear model, (1) (with the same parameters for males and females), rather than the sex-differentiated von Bertalanffy model chosen by De Leo & Gatto (1995). For each glass eel, a random number ε was generated, and the subsequent length of that individual at any age was calculated from equation (1). To save computing time, instead of allowing ε to vary continuously, it was picked at random from a set of 20 equi-probable numbers (the $i/40$ quantiles of the standard normal distribution, where $i = 1, 3, 5, \dots, 39$). Thus, all eels in the population fell into one of 20 growth cohorts, within each of which all eels followed the same length-at-age line. Seventy age classes were used, with the last being treated as a plus group.

Natural mortality was assumed to have the same value, M , for all ages and years. All eels greater than a minimum legal weight, W_{ML} , were assumed to be equally vulnerable to fishing.

The 16 model parameters are shown in Table 3.

Table 3: Model parameters and base-case values (see text for details)

Type	Parameter	Value
Growth	a_1	8.1 cm
	a_2	1.2 cm y^{-1}
	b_1	2.42 cm
	b_2	0.34 cm y^{-1}
Length-weight	α	1.48×10^{-3}
	β	3.08
Recruitment	σ_R	0.6
	ϱ_R	0.2
Natural mortality	M	0.04 y^{-1}
Minimum legal size	W_{ML}	220 g
		Male Female
Maturity	γ	0.7 0.25
	λ	40.6 cm 54.6 cm
	η	0.5 cm 2.8 cm

The annual cycle of the model was as follows.

1. The ages of all existing eels were incremented by 1 year (and their lengths and weights increased accordingly).
2. The number of glass eels (age = 0) for the year was generated and allocated equally amongst 40 groups (20 growth cohorts of each sex).
3. The proportion of each group that would mature that year was calculated (using (2)) and these eels were removed from the population.
4. The number of eels in each group was multiplied by e^{-M} to account for natural mortality.
5. For all groups that were of legal size, the number of eels was reduced by multiplying by $1-E$, where E is the exploitation rate.

In all simulations the mean number of glass eels per year was set so that the mean biomass of legal-size eels in an unfished population was 100 t.

5 The simulations

5.1 Interpreting the objective

Before carrying out the simulations it was necessary to make specific interpretations of two phrases in Objective 5. The first phrase is “determining the current status”. There are a number of ways in which this could be interpreted. The approach taken here was to simulate a fishery in which the biomass started to decline and to see

whether it was possible to detect that decline with at least 80% confidence. Thus, we are attempting to distinguish the stock status “biomass declining”, from the status “biomass stable”. More specifically, we are asking how steep must be the decline in biomass before we can be at least 80% confident of making this distinction. An associated question is “How many years will it take to confidently detect the decline?”. (All references to biomass in Section 5 mean the biomass of legal-size eels.)

The second phrase needing interpretation is “analysis of size frequency data”. We have taken the obvious interpretation and assumed that this means testing for trends in the mean length (or mean weight) or eels in the catch. It is clearly only statistics of location (e.g., mean, median, mode) that are of interest here, and the correspondence between our comparatively crude model and real fisheries does not seem good enough to be able to distinguish between trends in different location statistics. A significant trend in mean size (length or weight) is taken to indicate a trend in biomass. The procedure used for determining when an apparent trend in mean size was significantly different from zero is described below.

5.2 Simulation procedures

A set of 10 exploitation scenarios, each covering a 20-year period, was considered (Table 4). In all scenarios the exploitation rate, E , was fixed at 0.1 for years 1–10 to ensure the population was stabilised. In scenario S0, the only one where exploitation was stable, E was held at 0.1 for years 10–20; in scenarios S1–S6, E increased immediately in year 11 and stayed at the same level for the remaining years; in scenarios S7–S9, E increased gradually to 0.4 and then stayed at that level. Each scenario was simulated 500 times.

Table 4: Annual exploitation rates over a 20-year period for 10 scenarios used in simulations

Scenario	Year											
	1–10	11	12	13	14	15	16	17	18	19	20	
S0	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	
S1	0.10	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.15	
S2	0.10	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	
S3	0.10	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	
S4	0.10	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	
S5	0.10	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	
S6	0.10	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40	
S7	0.10	0.20	0.30	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40	
S8	0.10	0.15	0.20	0.25	0.30	0.35	0.40	0.40	0.40	0.40	0.40	
S9	0.10	0.13	0.16	0.19	0.22	0.25	0.28	0.31	0.34	0.37	0.40	

All scenarios except S0 cause the biomass to decline. For each one, the object was to determine how soon the decline in biomass could be reliably detected. That is, we wished to know what is the smallest number n for which we could be at least 80% confident that the n -year sequence of mean sizes starting at year 11 would show a significant decline. The decline was considered significant if $m < -m_n$ where m is the slope of the regression of mean size on time and m_n is the 95th percentile of the set of absolute values of the slopes from the analogous n -point regressions using mean sizes from S0. The reason that significance is determined by comparison with the S0 results, and not using the standard regression test, is that, as will be seen below, the

mean size data are strongly autocorrelated. This would make the standard significance tests too liberal (this is illustrated below).

Sampling error is an important issue in determining whether we can detect a trend in mean size. If the estimates of mean size from the shed-sampling programme have large errors then it will be much more difficult to detect a change in mean size. To start with, significance tests were calculated using just the simulated mean size data. These tests indicate what trends could be detected if there were no sampling error. Then the tests were repeated after adding normally distributed errors to the simulated mean size data. We took 1 cm and 30 g as “typical” standard errors for mean length and mean weight, respectively. Francis (in prep.) showed that for most catchments sampling error for the dominant eel species will be less than or equal to these values with moderate sampling effort. Initially, we intended to consider the effect of errors of 0.5, 1, and 2 times these values. However, so few trends were detectable at the typical error level (see below) that there seemed little point in considering twice that level.

The simulations were carried out in two stages: base case and sensitivity analysis. In the base-case simulations a single set of biological parameters was used; in the sensitivity analysis, alternative values of individual parameter values were considered to see which parameters were most influential, and how strong their influence was.

5.3 Base-case simulations

Base-case parameter values (Table 3) were (mostly) those for shortfin eels at Te Waihora. This is because this is the only species and location for which reliable information was available on length and age at maturity. The recruitment parameters were the only ones whose values were not derived from New Zealand data. These were taken as the median values in Figure 4.

Results from the base-case simulations with the stable scenario, S0, (Figure 6) illustrate some important features of all the simulation results. First, although, as expected, there is no long-term trend in length, weight, or biomass, each of these variables does show marked short-term variation. The problem is to detect long-term trends in the presence of this short-term variation. Second, mean size cannot be used to track short-term fluctuations in biomass. This is apparent in the lack of correlation in Figure 6D (the analogous plot with mean weight shows a similar lack of correlation).

A third point is that the mean size data are strongly autocorrelated. Over the 500 simulations, the median lag-1 autocorrelations were greater than 0.8 for both length and weight. A consequence of this is that the usual regression statistics are misleading. For example, over the 10-year period from year 11 to year 20, about 65% of the simulations would show a significant regression slope for mean length or mean weight against time if we used the usual t -statistic at the 5% significance level. This shows why we cannot use the usual regression significance test to determine whether a trend in mean size is significant. This problem is less marked, though still of concern, when sampling error is introduced: only 16–22% of these regressions were significant

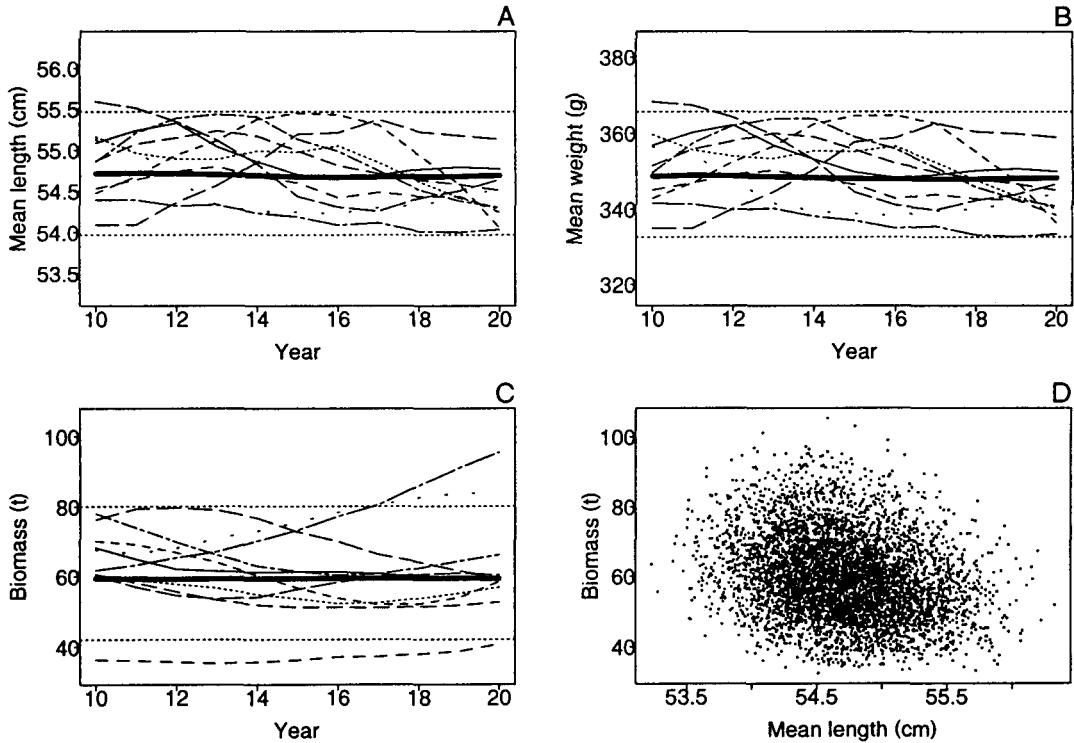


Figure 6: Some results from the base-case simulations with the stable scenario, S0 (exploitation rate fixed at 0.1). In panels A–C the heavy line is the mean over all 500 simulations; the horizontal dotted lines are 95% confidence limits for all simulations; and each of the 10 oscillating fine lines is the result from one simulation. Panel D shows mean length against biomass for years 11 to 20 in all 500 simulations.

when errors of half the typical size were included; and with typical size errors (1 cm or 30 g) this was reduced to 8–10%.

The average effect of the various non-stable scenarios is quite large in terms of biomass (with reductions of 20% to 70%), but relatively small in terms of mean length (reductions of 0.7 to 3.2 cm) and mean weight (16 to 67 g) (Figure 7).

Thus it is not surprising that the biomass decline was usually not reliably detected for typical sampling errors ($\varepsilon = 1$ cm or 30 g), and was never reliably detected for scenarios S1–S3, even when there was no sampling error ($\varepsilon = 0$) (Table 5). When it was detected, this was only after the biomass had already declined substantially (by 44 to 64% when there was no sampling error, and by over 60% with sampling error). Without sampling error, mean length and mean weight were equally useful in detecting a biomass decline; with typical errors the former was slightly better.

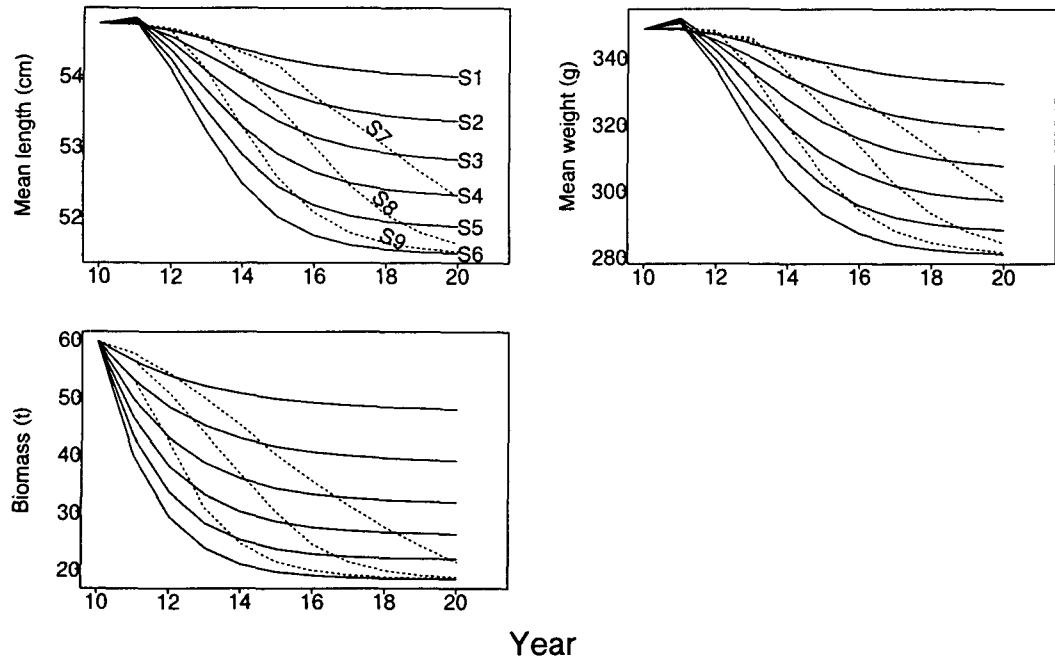


Figure 7: Mean results (averaged over all 500 simulations) for years 10 to 20 and scenarios S1–S9 (with the base-case parameter values): A, mean length; B, mean weight; and C, biomass. (The fact that mean size increases in year 11 for some scenarios and decreases for others is a rounding artefact).

Table 5: Results of significance tests on base-case simulation runs. Two numbers are given for each combination of scenario and sampling error, ε : the year in which the decline in biomass is first reliably detected (i.e., with at least 80 % confidence), and (in parentheses) the expected percentage reduction in biomass from year 10 to that year. – = decline not reliably detected

Scenario	Length			Weight		
	$\varepsilon = 0 \text{ cm}$	$\varepsilon = 0.5 \text{ cm}$	$\varepsilon = 1 \text{ cm}$	$\varepsilon = 0 \text{ g}$	$\varepsilon = 15 \text{ g}$	$\varepsilon = 30 \text{ g}$
S1	–	–	–	–	–	–
S2	–	–	–	–	–	–
S3	–	–	–	–	–	–
S4	13 (45)	–	–	13 (45)	–	–
S5	12 (44)	15 (61)	–	12 (44)	17 (62)	–
S6	12 (51)	14 (65)	–	12 (44)	17 (69)	–
S7	14 (59)	16 (67)	18 (69)	14 (51)	17 (68)	–
S8	17 (64)	18 (67)	20 (69)	17 (59)	19 (68)	–
S9	20 (64)	–	–	20 (64)	–	–

5.4 Sensitivity analysis simulations

The alternative parameter values used in the sensitivity analysis are shown in Table 6. Where possible, the alternative values represent extreme values from empirical data. Thus, slow and fast growth are represented by the parameter values for Waihao shortfin and Pigeon Bay Stream longfin, respectively (see Figure 2), and the alternative values for recruitment parameters are the extreme values in Figure 4. The alternative value for natural mortality is double the base-case value (with such a low value there didn't seem much point in considering a much lower value).

Table 6: The thirteen alternative sets of parameter values used in the sensitivity analysis, and a description of their effects. For each option the parameter values not specified are as in Table 3.

Alternative	Alternative values	Description of effect
A	$b_1 = 3.58 \text{ cm}$, $b_2 = 0.67 \text{ cm } y^{-1}$	Faster and more variable growth
B	$b_1 = 1.05 \text{ cm}$, $b_2 = 0.19 \text{ cm } y^{-1}$	Slower and less variable growth
C	$\sigma_R = 0.3$	Less variable recruitment
D	$\sigma_R = 1.0$	More variable recruitment
E	$\rho_R = -0.1$	Less correlated recruitment
F	$\rho_R = 0.6$	More correlated recruitment
G	$M = 0.08 \text{ } y^{-1}$	More natural mortality
H	$\gamma = 1 \text{ (M), } 0.5 \text{ (F)}$	Earlier migration, with less variable length
I	$\gamma = 0.5 \text{ (M), } 0.1 \text{ (F)}$	Later migration, with more variable length
J	$\lambda = 48.7 \text{ cm (M), } 65.5 \text{ cm (F)}$	Later migration
K	$\lambda = 32.5 \text{ cm (M), } 43.7 \text{ cm (F)}$	Earlier migration
L	$\nu = 0.5 \text{ cm (M), } 0.5 \text{ cm (F)}$	Less variable length at migration
M	$\nu = 5 \text{ cm (M), } 5 \text{ cm (F)}$	More variable length at migration

The effect of these alternative parameter values was gauged using just exploitation scenarios S4 and S8, and assumed sampling errors of 0.5 cm and 15 g. For each scenario and alternative we calculated the percentage confidence with which we could detect a biomass decline in a given year. That is, we calculated the percentage of simulations (out of 500) for which the regression of mean size (length or weight) on time was significant. The results were compared to the corresponding values from the base-case simulations.

The effect of the alternative parameter values was greater for scenario S4 than for S8, but was never large (Figure 8). The largest effect was usually caused by changes in the migration parameter, γ , (alternatives H and I). With all the migration parameters, changes that increased the range of the size distribution of the legal population made it easier to detect change (and vice versa). Changes to the parameter, ν , which has little effect on this range, changed the detection rate only slightly. As might be expected, increases (or decreases) in recruitment variability or correlation made it more difficult (easy) to detect changes in mean size.

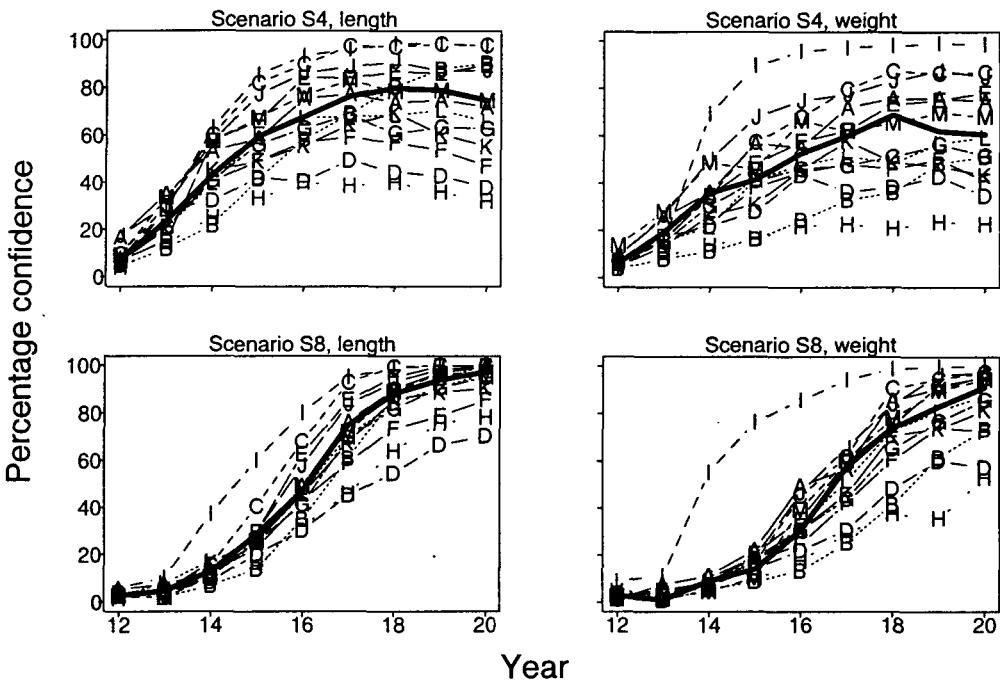


Figure 8: Results of sensitivity analyses. Plots of the percentage confidence with which a biomass decline could be detected against the year in which it is tested for, for the base case (heavy line) and for sets of alternative parameter values (light lines and plotting symbols as in Table 6).

6 Review of overseas eel management practice

Information was obtained through a search of recent publications and posting a note on an international bulletin board of eel researchers and eel fishery managers. Fortunately, a comprehensive review of the management of the European eel fishery has recently been published (Moriarty & Dekker 1997), and this provided useful information for Europe. Replies from colleagues in North America indicated that there is little interest in eel management there, but this has prompted them to hold a half day seminar on eel management as part of the 1998 American Fisheries Society meeting.

Six of the 11 European countries that maintain commercial eel fisheries have some form of size limit (Moriarty & Dekker 1997), either via a nominated minimum size of eels, or by way of minimum fyke net mesh size (Table 7).

Table 7: Summary of eel management practices in European countries

Country	Minimum mesh size	Other gear control	Closed seasons	Licences	Minimum size of eel
Sweden		✓	✓		Generally 55 cm
Denmark	✓	✓	✓		Yes but unknown
Ireland (N)	✓	✓	✓	✓	30 cm, but 41 cm preferred by industry
Ireland (R)	✓	✓	✓	✓	Yes but unknown
Netherlands	✓	✓	✓	✓	28 cm, but proposed to move to 32 cm
Italy	✓	✓	✓	✓	25 cm
Great Britain	✓	✓		✓	No, except 100 g in Thames
Germany		✓		✓	
France		✓	✓		
Portugal		✓	✓		
Spain		✓	✓	✓	

From the above and other literature reviewed, there was no evidence that any countries regularly review size distributions of harvested eels and then adjust regulations accordingly. There is a lack of management consistency between European countries, largely because regulations have evolved to suit local conditions and traditions — some measures which affect eels were originally implemented to manage other species or even navigation (Moriarty & Dekker 1997). Until 1984, eels were considered a nuisance species in France and were systematically removed from salmonid waters (Moriarty & Dekker 1997), a similar practice to that carried out in New Zealand during the 1930s and 40s (McDowall 1990).

7 Discussion

The main conclusion of this work is that estimates of mean size from the shed-sampling programme are of limited use, by themselves, in determining the status of eel stocks. Natural variations in glass eel recruitment cause substantial year-to-year variations in the mean size of eels, even when fishing pressure and stock status are constant. This makes it difficult to detect changes in mean eel size caused by long-term changes in fishing pressure. Even when there is no sampling error, biomass would need to drop by more than 40% (or more, if the drop is gradual) before this change could be reliably detected. The presence of sampling error at typical levels found by Francis (in prep.) makes this detection even harder. Thus, the mean size data are useful for detecting large long-term changes in stock status but not for year-to-year management.

Of course, these conclusions are contingent on the assumptions made in constructing the above model. There are many features of eel population dynamics for which we have evidence, but which are not represented in this model. Examples include:

- variation in the sex ratio of migrants (Vøllestad & Jonsson 1988);
- variations in size of glass eels (correlated with recruitment strength – Dekker 1996);
- density dependence (in survival, Vøllestad & Jonsson 1988; in survival, sex ratio, and age and length at migration, De Leo & Gatto 1996);

- sex determination of juvenile eels related to density (references in Vøllestad & Jonsson 1988; Parsons *et al.* 1977) or initial growth rate (Holmgren *et al.* 1997);
- annual variation in length (and age?) of migrants (Todd 1980)
- annual variation in growth rates
- variation in weight at length
- natural mortality as a (decreasing) function of age (De Leo & Gatto 1995)

It would not be difficult to include these in a population model. This was not done here either because there were insufficient data to allow determination of realistic parameter values for New Zealand eels (e.g., how fast does natural mortality decrease with decreasing density?), or because it seemed unlikely that their effect would be substantial. It is clearly unrealistic to assume that all eels of a given length have the same weight. However, the key relationship in the above model is weight-age, rather than weight-length, and variation in weight for a given age in the model was found to be comparable with that in the shed-sampling data.

For some of these features there is reason to believe that their inclusion in the model would make the results even more pessimistic. This is likely to be true for those that increase the extent of year-to-year variation in mean size (e.g., annual variation in growth or migration rates, or size or sex ratio of glass eels) or decrease the change in mean size associated with a given drop in biomass (e.g., density dependent growth rates). Nevertheless, there is still substantial uncertainty about many aspects of New Zealand eel population dynamics. For this reason the above results cannot be considered as definitive.

Another reason to believe that the real situation may be more pessimistic than is suggested here has to do with the statistical test used above to detect a change in mean size (see Section 5.2). We used information about fluctuations in mean size when there is no long-term trend in biomass to determine the significance threshold, $-m_n$. In practice, this information would not be available and it would be necessary to use some sort of regression procedure that took account of autocorrelated errors (e.g., Beach & MacKinnon 1978). Because of this our estimates of power must be interpreted as upper limits.

Fisher behaviour is another important factor that is unrepresented in the above model. We have assumed that all eels above the minimum legal size are equally vulnerable to capture. However, fishers react to changes in eel abundance and market demand by modifying their methods and locations of fishing. These modifications will affect the mean size of eels in their catch and thus make it more difficult to detect changes that are caused directly by increases or decreases of eel abundance.

It is worth pointing out that there are examples of fish species where substantial declines in biomass have not caused a corresponding change in mean size (Francis & Smith 1995). The reasons for this are unclear.

We have considered only one type of change in stock status: a long-term drop in biomass associated with an increase in fishing pressure. Such a drop could also be

caused by a long-term drop in mean recruitment, as appears to have occurred with European and American eels in the 1980s (Castonguay *et al.* 1994, Desaunay & Guerault 1997, Knights *et al.* 1996). It is obviously also important to be able to detect long-term increases in biomass. However, there is no reason to assume that these other types of change would be any easier (or harder) to detect than the type which we did model.

Another result from these simulations concerns the short-term changes in biomass and mean size that are caused by recruitment variation and are illustrated in Figure 6. The changes in biomass are not inferable from those in mean size because the two sorts of fluctuation are not in phase (thus the lack of correlation in Figure 6D).

Eel fisheries in Europe and North America are much older than those in New Zealand. However, we have not been able to find any references in the extensive literature on freshwater eels to the use of size data alone to monitor eel stocks. This is further reason to doubt that such monitoring is likely to be useful for year-to-year management.

What remains an open question is how useful these data would be in conjunction with the age data that are currently collected as part of the shed-sampling programme. There are many fisheries for which catch-at-age data are a key input to stock assessment — using virtual population analysis or other modelling approaches (e.g., Pope 1972, Doubleday 1976, Fournier & Archibald 1982, Megrey 1989). It is yet to be established whether such approaches could be useful for eel fisheries and, if so, what sort of age data should be collected (e.g., a full age distribution of the catch or simply enough data to estimate mean age at minimum legal size).

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Appendix 1: A review of published information on recruitment, growth, and migration data for New Zealand eels

Recruitment of glass-eels

Apart from some experimental glass-eel fishing during the 1970s, New Zealand has had no glass-eel fisheries and hence no time series of annual recruitment. This contrasts strongly with Europe where commercial catch data for glass-eels frequently cover 20+ years. The annual catches from the Waikato River experimental glass-eel fishing (Jellyman 1979) varied between successive years as much as 9-fold (Appendix Table 1).

Appendix Table 1: Catches of glass-eels from the Waikato River, 1970–74.

Year	Annual catch (kg)
1970	1 874
1972	2 066
1973	6 363
1974	708
Total	11 011

There are no records of effort expended over these years, although it is known that effort during 1974 exceeded that of previous years, as the target catch was about 10 t.

The total recruitment of juvenile eels (mainly age classes 0 and 1) into Lake Pounui was monitored annually (Jellyman & Ryan 1983). Results for 4 years (August – July) (Appendix Table 2) showed almost 9-fold variability in successive years (1974–75 vs 1975–76) and 28-fold between extremes (1974–75 vs 1976–77).

Appendix Table 2: Seasonal (August – July) catches of juvenile eels recruiting into Lake Pounui, Wairarapa.

Year	Annual recruitment
1974–75	6 754
1975–76	59 182
1976–77	190 009
1977–78	116 137
Total	372 082

The numbers of elvers recorded through hydro-dam elver passes, or manually transferred, have been reviewed by Beentjes *et al.* (1997). The most extensive time-series are for manual transfers at the Aniwhenua Dam on the Rangitaiki River, Bay of Plenty, where at least 6000 elvers were transferred in 1985–86, and a maximum of 144 500 in 1995–96; unfortunately there are no parallel records of effort, so it is uncertain whether these data indicate real differences in elver abundance.

Age and growth

There have been a number of studies on the age and growth of New Zealand eels. The most significant of these studies are summarised below (Appendix Table 3); not included are the extensive recent data generated as part of the Ministry of Fisheries catch sampling programmes (Beentjes & Chisnall 1998) which provide growth data

for an additional 20 and 13 sites for shortfins and longfins respectively in the North Island and 16 and 18 sites for these species the South Island.

Appendix Table 3: Review of data on age and growth of New Zealand freshwater eels. Parameters from age-length relationships (age = number of years in fresh water). sf = shortfin, lf = longfin

Location	Years	Sp.	No. aged	a	b	R ²	Reference
Waihola Channel, Taieri River	1973	lf ¹	44	195.2	16.8	0.77	Harries 1974
Henley, Taieri River	1973	lf ¹	23	222.0	25.7	0.83	Harries 1974
Meggat Burn, Taieri River	1973	lf ¹	49	130.0	15.2	0.81	Harries 1974
Silverstream, Taieri River	1973	lf ¹	42	183.6	25.0	0.83	Harries 1974
Lee Stream,	1973	lf ¹	50	145.1	15.2	0.83	Harries 1974
Sutton Creek, Taieri River	1973	lf ¹	63	139.3	18.5	0.94	Harries 1974
McGregors Drain, Taieri River	1973	lf ¹	49	138.1	26.3	0.88	Harries 1974
Puerua, Clutha River	1973	lf ¹	40	317.2	10.0	0.61	Harries 1974
Balclutha, Clutha River	1973	lf ¹	17	317.2	10.0	0.58	Harries 1974
Ahirau Stream, lower Waikato	1987–	lf ¹	29	58.7	35.8	0.80	Chisnall & Hicks 1993
Hakarimata pastoral streams, Lower Waikato	1987–	lf ¹	53	27.7	24.1	0.88	Chisnall & Hicks 1993
Mangapiko Stream, Lower Waikato	1987–	lf ¹	30	150.6	25.1	0.87	Chisnall & Hicks 1993
Hakarimata forested streams, Lower Waikato	1987–	lf ¹	22	68.2	14.7	0.86	Chisnall & Hicks 1993
Mangakara forested streams, Lower Waikato	1987–	lf ¹	11	249.9	11.9	0.93	Chisnall & Hicks 1993
Lake Karapiro	1988	lf ¹	62	318.7	19.4	0.57	Chisnall & Hicks 1993
Lake Matahina	1988	lf ¹	22	190.7	18.8	0.62	Chisnall & Hicks 1993
Lake Pounui	1979	lf ¹	344	123.52	19.27	0.86	unpublished data
Lake Rotoiti	1991	lf ¹	111	245.2	8.7	0.79	Jellyman 1995
Lake Waikare	1989	sf ¹	47	-14.77	46.77	0.67	Chisnall & Hayes 1991
Lake Waahi	1989	sf ¹	79	61.42	28.04	0.83	Chisnall & Hayes 1991
Lake Whangape	1989	sf ¹	96	127.48	23.30	0.77	Chisnall & Hayes 1991
Whangamarino Swamp	1989	sf ¹	53	182.80	17.75	0.59	Chisnall & Hayes 1991
Lake Pounui	1979	sf ¹	614	134.93	14.99	0.88	unpublished data
Pukepuke Lagoon	1972	sf ²	127	1.216	0.524	0.87	Jellyman 1979
Wainono Lagoon east	1997	sf ³	115	3.852	0.938	0.81	Jellyman & Sykes 1998
Wainono Lagoon west	1997	sf ³	132	4.375	0.719	0.78	Jellyman & Sykes 1998
Wainono canal	1997	sf ³	73	4.447	0.632	0.71	Jellyman & Sykes 1998
Waihao River	1997	sf ³	87	3.707	0.890	0.87	Jellyman & Sykes 1998
Coopers Lagoon	1997	sf ⁴	62				Jellyman & Beentjes 1998
Kaituna Lagoon, Te Waihora	1974–76	sf ³	611	4.346	0.600	0.68	Jellyman <i>et al.</i> 1995
Kaituna Lagoon, Te Waihora	1994	sf ³	264	4.347	0.662	0.55	Jellyman <i>et al.</i> 1995
Timberyard Point, Te Waihora	1974–75	sf ³	1437	4.483	0.538	0.67	Jellyman <i>et al.</i> 1995
Timberyard Point, Te Waihora	1995	sf ³	208	4.488	0.488	0.30	Jellyman <i>et al.</i> 1995

Notes: ¹ length (mm) = a + bage (y)

² $\log_{10}[\text{length (cm)}] = a + b\log_{10}[\text{age (y)}]$

³ $\log_e[\text{length (mm)} - 60] = a + b\log_e[\text{age (y)}]$

⁴ von Bertalanffy curve ($L_\infty = 1192 \text{ mm}$, $k = 0.057 \text{ y}^{-1}$, $t_0 = -3.37 \text{ y}$)

Cairns (1941) gave length at age graphs and values for a wide size range of both species. However, his data were from pooled samples taken throughout New Zealand

and are consequently of little use. Burnet (1969) presented linear regression values for annual growth increments from recaptured tagged eels of both species, and constructed growth curves (no regression values available) from otolith readings. Mitchell and Davis-Te Maire (1994, 1995) aged samples of longfinned eels from the Waiau catchment (Southland) and Lake Coleridge and associated lakes respectively; eels ranged in size from about 25 to 125 cm. Growth was expressed as a weight/age relationship (weight = g). i.e.,

$$\text{Waiau: } \log_e(\text{weight}) = 2.094 \log_e(\text{age}) - 0.480 \quad (R^2 = 0.55)$$

$$\text{Coleridge: } \log_e(\text{weight}) = 1.689 \log_e(\text{age}) + 0.990 \quad (R^2 = 0.32)$$

Size and age at seaward migration

Both size and age at migration are important parameters in development of a growth model. Appendix Table 4 provides a review of these data for New Zealand eels [migrating eels can be distinguished from others by their enlarged eyes and differences in coloration (Todd 1980)].

Shortfin males are the smallest of the migrating eels, and have a reasonably compressed size range of 338–598 mm, compared with 483–1090 mm for shortfin females. Both longfin sexes are larger than their shortfin counterparts, with males ranging from 482 to 736 mm, and females from 737 to 1560 mm. Trends in the mean age of migrants parallel those of size and are best demonstrated from the Makara Stream data where shortfin males and females average 14.2 and 19.4 years respectively, and longfin males and females 23.2 and 34.3 years respectively.

Appendix Table 4: Review of length (mm) and age (y) at migration of New Zealand freshwater eels. sf = shortfin, lf = longfin , M = male, F = female

Source	Sp.	Sex	Length				Age			
			No	Mean	SD	Range	No	Mean	SD	Range
1	sf	M	65	465	43	381-598	35	14.2	4.34	6-23
1	sf	F	34	737	93	563-933	18	19.4	4.94	10-30
1	lf	M	362	623	46	482-736	158	23.2	3.88	12-35
1	lf	F	22	1063	164	781-1333	15	34.3	6.14	25-48
2	sf	M	120	447	37.1	370-549	79	14.3	2.76	8-22
2	sf	F	47	764	104.2	550-1023	43	22.5	5.25	12-35
3	sf	M	12020	432	22.2	338-554	1263	14.4	2.0	9-24
3	sf	F	778	609	92.8	483-1024	181	23.6	3.59	13-35
3	lf	M	12	666	65.5	558-735	-	-	-	-
3	lf	F	176	1156	174.3	737-1560	5	49.4	-	25-60
4	sf	M	914	392	23.2	335-485	163	14.4	2.0	9-22
5	sf	M	101	413	34	352-525	49	21.9	4.1	14-33
5	sf	F	170	741	113	567-1090	78	40.8	7.3	26-60

- Sources: 1. Makara Stream, 1972–73 (Todd 1980);
 2. Lake Onoke, 1971–72 (Todd 1980);
 3. Te Waihora, 1972–80 (Todd 1980);
 4. Te Waihora, 1993–96 (Jellyman & Todd in press);
 5. Lake Pounui, 1976–78 (unpublished data)