

Biological data and model development for management of longfinned eels

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Non-technical Summary

1998/128	Biological data and model development for management of longfinned eel fisheries
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Objectives:

1. Estimate population parameters required for a management model. These include survival, density, age structure, growth, age and size at maturity and at recruitment to the adult eel fishery. Estimate their variability among individuals in a range of habitats.
2. Develop a management population dynamics model and use it to investigate management options.
3. Establish baseline data and sustainability indicators for long-term monitoring.
4. Assess the applicability of the above techniques to other eel fisheries in Australia, in collaboration with NSW. Distribute developed tools via the Australia and New Zealand Eel Reference Group.

<p>Outcomes achieved: As a result of the development of the management model in this project the current management arrangements will not alter substantially in NSW and Queensland. Both states will continue to persist with areas closed to fishing. An agreement between DEH and DPI&F substantially recognises most of the findings and recommendations from this project, including using long-term monitoring CPUE data as an index of abundance and sustainability indicator and maintaining closures in freshwater riverine areas. Both DEH and DPI&F acknowledge the importance of passage/trap and transport of elvers past artificial barriers and cost benefit options of different mechanisms are likely to be investigated in the future.</p>

This project developed a user-friendly eel fishery management model to enable fisheries managers to investigate different management alternatives and their likely effects on trends in yield to the fishery and sustainable production of spawners.

In order to develop the model it was first necessary to validate whether eels from sub-tropical Queensland could be aged. The results of ageing validation experiments on both tank held eels and on tagged wild eels confirmed that long-finned eel stocks in sub-tropical Queensland could be aged reliably. Confirmation that eels could be aged, enabled estimation of parameters such as age at maturity, growth, mortality and age at recruitment to the fishery. These parameters and others such as selectivity of the eel fishery and size at maturity data enabled development of an eel fishery management model.

Male eels in Queensland were observed to mature between 4 and 17 years and female eels between 5 and 39 years. Long-lived species such as eels that only spawn once are particularly vulnerable to overexploitation. Results of modelling suggest that long-finned eel stocks in Australia are best managed by having substantial areas closed to the fishery, to enable maturation and escapement of spawning stock. The eel fisheries in Queensland and NSW are already essentially run along these lines, with freshwater riverine areas excluded from the fishery in both states.

Within Queensland the fishery is confined to impoundments and farm dams, and in NSW the fishery is concentrated in estuarine areas with some fishing also permitted in impoundments and farm dams. The results of the modelling from this project are likely to ensure the current management arrangements continue in both states, with no expansion of the fishery into freshwater riverine areas.

Results of modelling also suggest that spawner production and productivity of the fishery itself is likely to be improved through provision of elver passes or other means that will increase access of eels past barriers such as high dams and weirs. Improvement of fish passage is one area where current management arrangements of fisheries in both states could be improved, both for the benefit of the fisheries and for improved sustainability.

To monitor effectiveness of current management arrangements in sustaining production of spawners it has been recommended that the DPI&F freshwater fisheries long-term monitoring program be continued. An evaluation of this program showed that fishery independent CPUE data obtained from surveys of riverine areas was suitable for use as an index of abundance and sustainability indicator. Data collected by the program will be sufficient to detect changes between rivers and between years and is also able to be balanced for the influence of habitat parameters on CPUE.

Keywords: *Anguilla*, longfinned eel, fishery model, eel fisheries.

Introduction

Background

This study supports the development and sustainability of the eel aquaculture industry, helps management of adult eel fisheries, and helps maintain the integrity of freshwater ecosystems in Queensland and throughout Australia. The research provides a management model for long-finned eels and the data to support the model in Queensland. It develops the data structures and analysis protocols to allow the model to be distributed and used for all Australia's eel stocks, both short-finned and long-finned. The research also develops methodology and data structures for a fishery-independent sustainability indicator suitable for eel stocks throughout Australia, and collects baseline data for monitoring sustainability in Queensland.

The proposal was developed through discussions with the eel aquaculture industry and adult eel fishery representatives, glass eel researchers, and international eel experts. It arose from a perception of great potential risks of fishery collapse and very little knowledge of the long-finned eel. Perceptions were supported by preliminary population modelling work. The project had the support of the Queensland Fishery Service and industry groups. We consulted with the Freshwater Management Advisory Committee and eel aquaculturists. We also consulted with Drs Rick Fletcher, Bruce Pease, and Kevin Rowling of New South Wales Fisheries, who carried out a complementary and collaborative research program for New South Wales.

There are four species of Anguillid eel in Australia, of which two are common: the long-finned eel (*Anguilla reinhardtii* Steindachner, 1867) and one of the three short-finned species (*A. australis australis* Richardson, 1941). Eels are catadromous, migrating huge distances to spawn. Australian long-finned eels are thought to spawn in the Coral Sea. The East Australian Current carries the larvae down the coast, where they metamorphose and enter estuaries anywhere from Cape York to Tasmania. In general, female eels grow larger and mature much later than males, and are found more often in the upper catchment than in estuaries. Males are seldom found outside brackish and estuarine areas but are known to occur in freshwater even in upstream areas.

Industry size and potential

Glass eel fisheries support a high value aquaculture industry worth well over \$1 billion in Asia and Europe. Japan alone consumed over 100 000 tonnes of eel in 1996, and Europe uses about 25 000 tonnes per year. The growth of the Chinese and other Asian economies, where eel products have high value as a prestige food item, suggests that the value of this industry will continue to increase.

At the same time, supplies of glass eels from traditional sources in Asia and Europe are declining. Since the early 1980s German catches have dropped by 90 per cent and French by 86 per cent. American, Taiwanese and Japanese eel supplies have also dropped substantially. There are now severe shortfalls in domestic supplies of glass eels in Japan and Taiwan, leading to very high prices and the rapid development of additional fisheries in, for example, the USA (New York Times, 16/2/1997). Catches there have also fallen. These drastic declines offer Australia both an opportunity and a warning.

Queensland is well placed to take advantage of this opportunity. The two main species of glass eels caught in our waters (*Anguilla australis* and *A. reinhardtii*) have good aquaculture potential and market acceptance. *A. australis* is potentially well accepted in the Japanese and Taiwanese markets, and *A. reinhardtii* in the Chinese market (Manhattan Industries, personal communication). An eel aquaculture industry is currently developing in Queensland, with a multi-million dollar facility recently completed in south-eastern Queensland. Other states also have multi-million dollar facilities. The industry was supported by an FRDC-funded project to identify the current status of and availability of glass eel supplies. An adult fishery also exists in Queensland, though its continued existence is under threat. The fishery has recently declined dramatically, apparently due to depletion of stocks in available waterways.

The Girramay, Gulmay and Jirrbal people traditionally harvest eels from the Tully-Murray area of Queensland by hook and line and basket traps. The Jumbin community (Girramay people) have expressed concerns about a reduction in eel catches on the Murray floodplain. This may be attributed to loss of wetland habitat. Nevertheless, commercial harvest of eels is also of concern to these people and the QFMA recommended that traditional use of eel resources needs to be considered in future management arrangements for the eel fishery (QFMA 1996).

Sustainability

The aquaculture industry depends on a continuing supply of glass eels. The FRDC glass eel project addressed supply identification and industry development, but did not address the overall sustainability of this very vulnerable resource. The growing value of eel products poses a serious threat to wild adult eel populations, which are the source of the glass eels. Adult eel fisheries have already declined in Queensland and New South Wales. High prices for adult eel products have encouraged expansion of the adult fishery into new areas as well as glass eel fishing, so the populations are doubly threatened. The risks involved are clear both from both the international declines, and the demonstrated ability of Queensland fishers to quickly deplete adult stocks (QFMA 1996) in the areas they fish.

Any fishing of adult eels has the potential to affect the glass eel supply. Eels are peculiarly vulnerable to fishing because of their unusual life history. Eels are very different from other fish, and cannot be managed in the same way. They have a very long generation time (approximately 20 years for Victorian shortfinned eels *A. australis* females, though up to 93 years has been documented for New Zealand longfinned eels *A. dieffenbachii* (Jellyman 1995)), low natural mortality, and all fishing occurs before breeding. Preliminary modelling suggested that even a low fishing mortality of 0.1 may reduce the number of female spawners produced from a stock to 23% of unfished levels. Since much of the fishery's potential value is generated by spawning females, the risks are clear. These estimates are based on data for *A. australis* in Victoria. Available evidence suggested the reduction may be more severe for *A. reinhardtii*.

Recently dramatic declines have occurred in catch rates of adult eels in Queensland (Teddy Paul, Manhattan Eels and Seafoods, secretary of Eel Fishers and Exporters Association, personal communication). For example, in Wivenhoe Dam 30 tonnes were caught in the 1995–96 season, 10 tonnes in the 1996–97 season, and 1.7 tonnes during the major catching period of the 1997–98 season. Fishers suggest that this may partly be a result of drought

leading to poor recruitment, but given the longevity and catchability of eels, overfishing compounded by poor recruitment is possible. Catches of longfinned eels in New South Wales have declined by 50% initially but have stabilised since the late 1990s.

The Queensland and NSW fisheries are very different. The Queensland fishery has operated mainly on females in fresh impounded waters, while the New South Wales fishery operates mainly on males in estuarine areas, although there is also an impoundment fishery component.

Research summary

Management of eel stocks requires a different paradigm from most other fisheries, due to their remarkable life history. Population modelling proposed in this project will provide a basis for sustainable management of eel stocks throughout Australia. The model or models will be adaptable for management of both eel species throughout Australia.

Very little is known about Queensland's eel stocks. Important demographic factors such as survival rates, age at maturity, and growth rate have not been studied for either of the major species in Queensland. Research on long-finned eels elsewhere in Australia has been very limited. Such research is clearly essential for informed management. We propose research into demographic factors important for management modelling of the longfinned eel fishery in Queensland and throughout Australia. The research framework, including database structures and a strong statistical component, will provide a basis for work on other eel stocks.

Monitoring of adults can provide early warning of declines and an indication of future glass eel trends, with far lower variability than glass eel monitoring. This project will develop the methodology and a database for collecting, analysing and sharing information on adult eel status and resource sustainability. It will develop the database structures, sampling techniques and statistical models to be used in ongoing monitoring.

Need

The research provides a management model for longfinned eels, and the data to support the model in Queensland. Supporting data for NSW were supplied by a collaborative project in that state. The model will also be suitable for managing shortfin eels in Victoria, Tasmania, NSW and Queensland, given appropriate data. The research has also developed methodology for a fishery-independent sustainability indicator, which will similarly be useful for both longfinned and shortfinned eels.

Glass eel fishing and the aquaculture it supports are developing industries throughout southern and eastern Australia, and are potentially worth tens of millions of dollars. Prices for adult eels have also increased in recent years, encouraging the growth of this industry. However, adult stocks in Queensland and NSW appear to be declining. In addition, eels comprise a major part of stream biomass, and are probably the most important predators in many freshwater ecosystems. Significant changes to stream ecology by major reductions in eel biomass have the potential to destabilise ecosystems, facilitating invasion by exotics among other problems.

The FRDC is supporting glass eel industry development. However, sustainability of glass and adult eel fishing is not yet being addressed. Internationally, eel fisheries have not been sustained. Glass eel supplies have collapsed in Europe, Asia, and North America.

Our preliminary modelling of Queensland eel stocks demonstrated two things. Firstly, fishing of adult eels can severely reduce the number of spawning females. This is backed up by evidence from New Zealand, where the Lake Ellesmere eel fishery has seen drastic declines in the number of (particularly female) spawners (Jellyman 1995). Thus some types of adult eel fishing may damage the glass eel fishery. On the other hand, reduced or redirected adult eel fishing may significantly enhance the glass eel fishery. A management model was required to provide insight into these issues. Modelling of this kind has not previously been published for eels, and interest has been expressed by international eel researchers.

Secondly, very little was known about longfinned eel demography and population structure, knowledge which is needed for informed management of eel stocks. Some very sparse demographic data come from New Zealand, Tasmania and Victoria, but even this is compromised by eels' great variability in growth and maturation rates between environments. Queensland may hold the majority of longfinned eel biomass in Australia, but no studies had been carried out either here or in NSW. Statistically sound fishery-independent techniques are required to estimate population structure and demography for all important sectors of the population, particularly females. Fishery-dependent techniques will not work in Queensland due to the decline of the fishery. Data from NSW will provide complementary information on males which, it was thought, are probably seldom found outside estuaries.

As the glass eel fishery develops and as demand for adult eels rises, information on the changing status of wild eel stocks will be required. A sustainability indicator can provide this. Such indicators are best developed as early as possible in the evolution of the fishery.

Eel life histories are complex and unique, and successful management requires a different approach from other fisheries. Successful management of glass and adult eel fisheries requires a management model supported by demographic and fishery-based data. It also requires a feedback mechanism in the form of a sustainability indicator. The proposed research will provide the first and develop methodology for the second.

Objectives

1. Estimate population parameters required for a management model. These include survival, density, age structure, growth, age and size at maturity and at recruitment to the adult eel fishery. Estimate their variability among individuals in a range of habitats.
2. Develop a management population dynamics model and use it to investigate management options.
3. Establish baseline data and sustainability indicators for long-term monitoring.
4. Assess the applicability of the above techniques to other eel fisheries in Australia, in collaboration with NSW. Distribute developed tools via the Australia and New Zealand Eel Reference Group.

Chapter 1: An index of abundance for adult eels

Simon Hoyle, Michael Hutchison and David Mayer

Abstract

The low catch rates in the four large dams surveyed suggest that densities of eels are too low in most dams for monitoring of reservoirs by electro fishing to be of much value in assessing trends in eels stocks in the region. The DPI&F long-term monitoring program (LTMP) was found to be a suitable method for evaluating trends in eel stocks in rivers. LTMP surveys can be used to generate an index of abundance for legal sized long finned eel stocks. Should downward trends be detected in electrofishing CPUE over several years, then such a decline could be used as an early warning signal. This could lead to activation of surveys along the lines of the south-east Queensland (SEQ) eel surveys. The SEQ eel surveys covered a wider range of habitats and were at a finer scale and may be more useful in confirming recruitment of smaller eels.

A number of environmental parameters were found to have a significant influence on river and stream catches of eels. The depth effect is most pronounced for small eels, with catches of small eels tending to be greater in shallow waters, including riffle habitats. Other habitat variables with significant positive effects on catch rates of both large and small eels were rocks and aquatic macrophytes. Large eels were found to be associated with snags (woody debris) and undercut banks.

Objective

Establish baseline data and sustainability indicators for long-term monitoring.

Introduction

Australian longfinned eels (*Anguilla reinhardtii*) are believed to belong to a single panmictic stock. Therefore the number of new recruits entering a river system in one part of eastern Australia (e.g. Southern NSW) depends on the number of spawners contributed from all other parts of the east coast, including SEQ. Monitoring for sustainability across the panmictic population should target recruitment of new eels into river systems across eastern Australia, and provide an index of abundance of adult eels. Monitoring glass eel arrivals would be designed to pick up overall changes in the level of recruitment, due to for example, a decline in spawning biomass. Local changes at a representative sample of sites would be informative about overall changes in the panmictic population. However, glass eel arrivals are weather-dependent (Chisnall *et al.*, 2002), and extremely variable in space and time (Pease *et al.*, 2003), particularly in Australian conditions of high rainfall variation. It is very difficult to monitor them at a level sufficient to detect long-term changes.

Monitoring adult eels is preferable for several reasons. Sampling to monitor river fauna is already underway in Queensland. Adult eels are quite easy to catch and their numbers are relatively stable on short time scales, due to their longevity. Long-term changes are therefore easier to detect. Such an index could also pick up local changes within rivers, which may occur as a result of the erection or removal of downstream barriers or other local factors such as changes to habitat condition or overfishing. Severe local impacts across a number of catchments could lead to eventual panmictic declines in recruitment. Therefore being able to identify declines within a river system is also important.

Having an index of abundance monitoring system in place will assist in early detection of declines, which will lead to research and management actions to identify and address the problem.

Methods

This project evaluated two possible methods for long-term monitoring of eel stocks using an index of abundance. The first is the survey method developed for the current project for collection of eels for population age structure data. The second is the Queensland Fisheries Service's Long-Term Monitoring Program for freshwater fish stocks.

Survey Methods: South-east Queensland eel surveys (this project)

Sampling concentrated on freshwater reaches since the females tended to concentrate in these areas. Female population dynamics are more sensitive to fishing and are more important to the sustainability of glass eel and adult fisheries. Sampling was restricted to south-eastern Queensland, with Bundaberg the northern limit. Sampling was stratified into reaches in small streams, reaches in large streams, and impounded waters. Impounded waters included three sites on Wivenhoe dam (fished), three sites on Lenthalls Dam (fished) three sites in Enoggera Reservoir (unfished), and three sites at Tingalpa reservoir (unfished). In addition ten sites were selected in small streams that had not been fished (tributaries of the Albert, Logan, Coomera and South Pine Rivers), ten sites in larger unfished waterways (Albert, Logan, Coomera and South Pine Rivers), and ten sites in larger fished waterways (Mary and Caboolture Rivers, Tinana Creek). Sites on two of the unfished streams (Albert and Logan Rivers) had glass eel fishing at the mouth, and glass eel sampling occurring under FRDC project 97/312. At all sites both longfinned and shortfinned eels were sampled. Locations of sampling sites are shown in Figure 1.

Within these sites, transects were chosen as randomly as practicable. Each waterway was divided into 10 by 10 km blocks, and three blocks selected at random. Within these blocks, all suitable access points were determined, and one of these was randomly selected as an appropriate starting point.

For each site distance from the sea and existence of barriers to migration such as barrages were recorded. At a 5 metre shot level (see below), depth and habitat variables were recorded. Habitat variables recorded included gross habitat type (pool, riffle, run, cascade and backwater), depth and width of stream. Other habitat features were scored on a semi-quantitative or categorical five point system of zero to four. Zero denoted the habitat feature was absent, whereas scores from 1 through to 4 denoted a range of importance from minor to major. Habitat features scored in this way were snags or large woody debris, undercut banks, aquatic macrophytes, aquatic grasses, rocks, overhanging vegetation, canopy cover, and side creeks.

At each site the size structure of the population was investigated by electrofishing along one hundred-metre transects. Each transect was broken up into 5 metre shots and fished from downstream to upstream. In large streams each of three transects was electrofished as separate 5 metre shots on each bank.

On small streams six transects were fished in series with both banks included in a single shot. In impoundments 5 metre shots were conducted along a randomly selected shoreline. Up to six transects were run in each impoundment.

All sites were fished in two passes. In small streams and shallow riffle habitats backpack electrofishing was used. In all other areas boat mounted electrofishing was used. It was assumed the efficiency of both electrofishing methods was approximately the same between these different environments. The length and sex of each eel obtained was recorded.

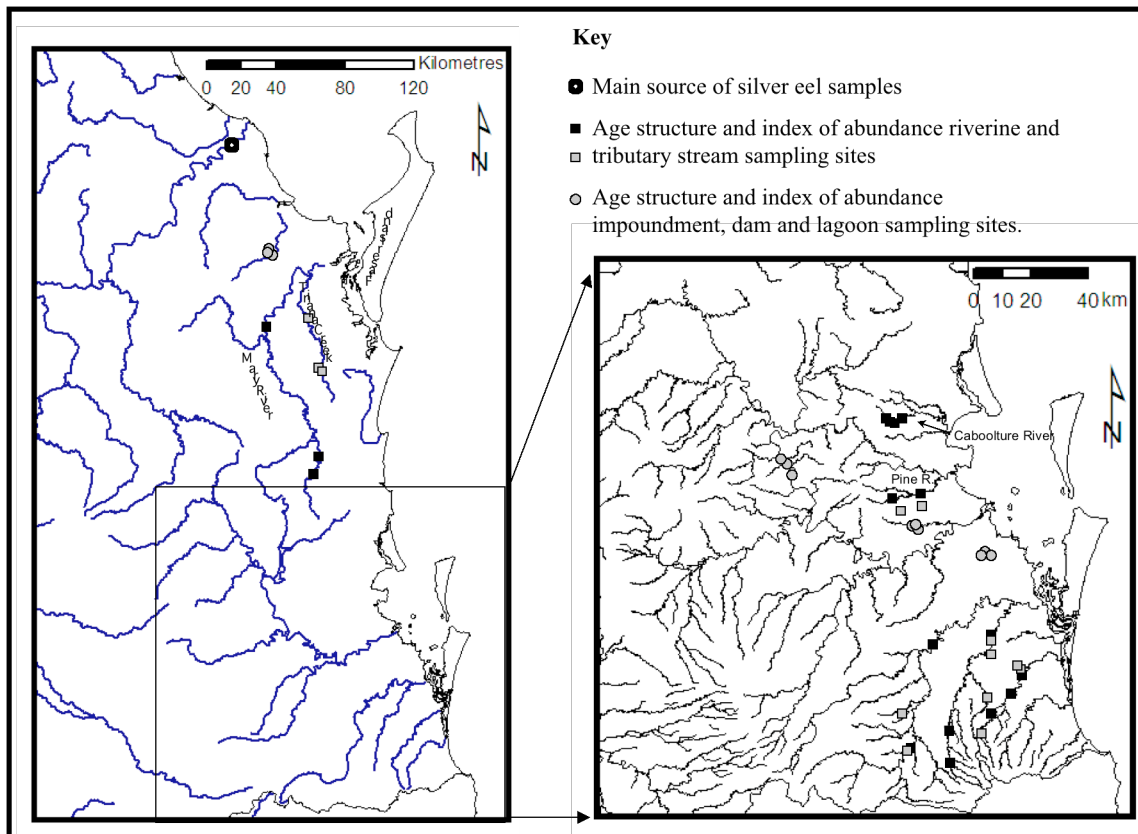


Figure 1: South-east Queensland eel survey age structure and index of abundance sampling sites.

Survey methods: Queensland Fisheries Service’s Long-Term Monitoring Program (LTMP)

The Queensland Fisheries Service Long-term monitoring program monitors ten river systems in Queensland by boat electrofishing on an annual basis. Six of the rivers are within the distribution of longfinned eels. Three are located in the southeast (Noosa, Albert-Logan and Mary River systems) and three are located in the tropical north (Herbert, Johnstone and Daintree Rivers). Eels comprise one of the target species for surveys within these rivers. Full descriptions of the methodology used for the long-term monitoring program are found in Hutchison *et al.* (2004). The methods are summarised here. On each of the rivers seven fixed reaches are sampled by electrofishing boat. Initial selection of reaches was random. Within each reach, six–50 metre sites or shots are selected randomly each year. Shot sites extend 15 metres from the bank. These sites are fished using standardised electrofishing boat manoeuvres involving pre-determined application of electrofishing power by time and space within the site. Numbers and total length of all eels captured is recorded at each site. At each of these 50 metre sites various habitat parameters are also recorded. Protocol for recording for habitat details was adapted from the system used by Russell *et al* (2000). Habitat details recorded included pH, conductivity (ms/sec), oxygen (mg/L), turbidity (NTU and Secchi depth cm), water temperature (°C), maximum depth (m), stream width (m), current velocity, bottom substrate, in-stream snags, rocks, grasses, leaf-litter, aquatic macrophytes, emergent vegetation, riparian vegetation width (m), riparian continuity and composition, canopy cover, cover, and index of disturbance.

The index of disturbance classifies a site as having extreme disturbance, high disturbance, moderate disturbance, low disturbance or undisturbed. Disturbance ratings are based criteria relating to adjacent land uses and riparian zone and in-stream indicators. Details of the criteria are in Hutchison *et al*, (2004). Scoring for other habitat criteria are as follows. Riparian continuity is classed as continuous, <25% breaks, <50% breaks, >50% breaks. Riparian composition is scored by

the per cent composition of trees, grasses and bare ground. In-stream habitat features such as rocks, snags, aquatic macrophytes, and leaf litter are scored from 0 to 5 based on the surface area of the shot zone each of these features occupies. A score 0 indicates the feature is absent, 1 = $<50 \text{ m}^2$, 2 = $50\text{--}100 \text{ m}^2$, 3 = $>100 \text{ m}^2$, 4 = $>150 \text{ m}^2$ and 5 = $>200 \text{ m}^2$. Overhead or canopy cover was scored in the same way. Undercut banks and emergent vegetation (reeds, rushes) were also scored from 0 to 5. In this case the score relates to the length of shoreline or bank of this type. If the feature is absent it is scored 0. A score of 1 = $\leq 10 \text{ m}$, 2 = $\leq 20 \text{ m}$, 3 = $\leq 30 \text{ m}$, 4 = $\leq 40 \text{ m}$ and 5 = $\leq 50 \text{ m}$. Bottom substrate was classed as boulder/cobble, cobble/gravel, sand and fine material. Each of these substrates was scored from 1 to 4, with 1 being the most abundant and 4 the least abundant.

Statistical Methods — south-east Queensland eel surveys

The survey design generally consisted of 20 shots within each site, at one sampling date. At each site, two successive samples were conducted, with the eels from the first 20 shots not being returned prior to the second shots being taken (this contributed better independence between samples). The total number of shots was 230 in dams and reservoirs, and 1270 in rivers and streams. True spatial or temporal replication of sites was minimal (6-degrees-of-freedom). However, the environmental variables of interest were measured within-sites, so shots is the appropriate experimental unit for fitting these terms, provided that any spatial autocorrelations amongst the residuals are low.

‘Dam’ and ‘other’ (riverine) sites clearly had different types of environmental variables, as well as different catch rates. Hence, these data sets were analysed separately. Sites with zero captures (over all times and shots), and hence zero deviances, were noted and excluded from these analyses.

A Poisson generalised linear model (GLM) with log link (McCullagh and Nelder, 1989) was used to model the numbers of eels, via GenStat (2000) — separately for total, ‘large’ ($>300 \text{ mm}$), and ‘small’ ($\leq 300 \text{ mm}$) eels. ‘Sample’ was fitted first (to estimate the fish-down effect), and was always significant ($P < 0.01$). Then, step-forward selection of main effects was employed to arrive at a model for total eels where all environmental effects were significant (at $P < 0.05$, and using the theoretical dispersion coefficient of one). Next, the ‘site’ term was checked for significance, to see whether the observed differences in eel numbers had been adequately explained by the environmental variables. Finally, all two-way interactions between these significant effects were tested for significance, and the tabulated means were checked for biological meaning. Residuals from this model were examined for spatial and temporal (between-samples) correlations, and adjusted means for each effect were tabulated.

A follow-up comparison between rivers with different fishing pressures (none, fished, glass-eel harvests) was also undertaken. Because this factor was totally aliased with ‘site’ (in that sites were nested within this classification), this ‘fishing pressure’ term was fitted as a 2-degrees-of-freedom contrast within the ‘site’ coefficients.

Hierarchical generalised linear models (Lee and Nelder, 2001) were also investigated for the riverine data (total eels), using GenStat (2000). These HGLM’s extend generalised linear models to allow the inclusion of an extra random term in this case, the top (but minimally estimable) level of replication. This term was modelled as a gamma distribution with the log link (Nelder, pers. comm., 2001). The Choleski method for adjusting the likelihood profile, and first-order Laplace approximation, were required for convergence.

Statistical methods: LTMP surveys

A Poisson generalised linear model (GLM) with log link (McCullagh and Nelder, 1989) was used to model the numbers of eels, via GenStat (2000) separately for total, 'large' (>300 mm), and 'small' (≤ 300 mm) eels. Step-forward selection of main effects was employed to arrive at a model for total eels where all environmental effects were significant (at $P < 0.05$, and with the more conservative approach of using the fitted dispersion coefficient). Next, the 'river' term was checked for significance, to see whether the observed differences in eel numbers had been adequately explained by the environmental variables. Finally, the 'year' term and 'year by river' interactions were added, to test for variations in the annual patterns between the rivers. Residuals from this model were examined for spatial correlations (between-shots, as there were generally six shots per sampling date), and adjusted means for each effect were tabulated.

Results

South-east Queensland eel surveys

In the hierarchical generalised linear model, the fitted deviance for the additional (top-level) replication was less than that of the full model. Hence, this HGLM defaulted back to the Poisson GLM. This pattern was also noted in an approximate split-plot GLM of $\ln(Y+1)$ -transformed data using the Normal distribution, where the top-level residual mean square was less than that of the main model. This further justifies using shots as the experimental units in the final Poisson GLM.

Dams Data

No eels were caught in Tingalpa Reservoir or Wivenhoe Dam. A total of 34 eels were caught at four sites in Enoggera Reservoir (3) and Lenthalls Dam (31). These were all classified as large; no smalls were captured.

Aquatic macrophyte was the only environmental variable to be significantly associated with capture rates. Unfortunately, it was highly aliased with site (also significant), making interpretation of these patterns (Table A1) somewhat difficult. This model explained 55% of the total deviance, and both the temporal and lag-one spatial correlations between model residuals ($r = -0.15$ and -0.08 , respectively) were not significant ($P > 0.05$). The fish-down effect was quite pronounced, with 76 per cent being captured in the first sample (adjust overall mean of 0.413, versus 0.127 for the second sample). The model was under-dispersed with a residual mean deviance of 0.61 — this indicates that these large eels were distributed more evenly than was expected by the random Poisson model, i.e., statistically, they displayed territorial behaviour.

Table A1. Adjusted average capture rates of large eels in dams, by site and type of aquatic macrophyte.

Dam	Site number	Aquatic macrophyte	Number of shots	Average no. eels (/shot)	Standard error
Enoggera Reservoir	1	4	20	0.10	0.07
	2	4	20	0.05	0.05
Lenthalls Dam	4	3	4	0.00	0.00
	4	4	16	0.31	0.14
	5	0	14	1.21	0.29
	5	1	6	1.50	0.50

Riverine Data

Only one site had zero captures Emery Bridge, on the Mary River, with 40 shots. For the other sites, table A2 summarises the goodness of fit for the Poisson GLMs. In every model, ‘site’ was a significant contributor after the environmental variables had been screened, indicating that the site differences could not be totally explained by these attributes. Tables A3 to A9 list the adjusted means for the significant terms in these models, namely depth, type of stream, rocks category, aquatic macrophyte, snags, undercut bank and overhanging vegetation. Interestingly, the non-significant variables were turbidity (Secchi method), width of stream (likely to be correlated with depth), grass, and canopy cover (likely to be correlated with overhanging vegetation). These either had no association with catch rates, or alternately may be correlated with other variables already in the model (and hence had no further statistical contribution). Tables A10 and A11 give the adjusted means for the sample and site effects, respectively.

Tables A12 and A13 show the adjusted means and raw means for the follow up comparison between rivers with different fishing pressures. In no cases was fishing pressure significant ($P > 0.05$), indicating that the observed ‘site’ differences could not be attributed to differing fishing pressures. However, the fitted means are of interest in that the mean values for fished sites are lower than for those of unfished sites, with the signal being quite strong (magnitude of 10) and close to significant for small eels, but the pronounced background variation means this is not a significant signal ($p = 0.07$).

Table A2. Goodness of fit of the Poisson GLMs for eel captures.

Parameter	Total count	Large eels	Small eels
Residual mean deviance	0.80	0.53	0.55
Deviance explained (%)	67.8	41.9	73.0
Spatial autocorrelation (r)	0.16	0.06	0.17
Temporal correlation (r)	0.12	0.14	0.07

Table A3. Adjusted average capture rates (per shot), by depth.

Depth	Total count	Large eels	Small eels
1	0.82	0.19	0.58
10	0.73	0.19	0.47
50	0.43	0.19	0.18
100	0.22	0.19	0.05
150	0.12	0.19	0.02
200	0.06	0.20	0.00
250	0.03	0.20	0.00

Table A4. Adjusted average capture rates (per shot), by stream physical habitat type.

Stream type	No. of shots	Total count	Large eels	Small eels
Pool	560	0.30	0.16	0.11
Riffle	180	0.60	0.22	0.24
Run	471	0.43	0.20	0.16
Cascade	8	0.93	0.71	0.33
Backwater	6	0.83	0.00	0.62

Table A5. Adjusted average capture rates (per shot), by rocks.

Rocks	No. of shots	Total count	Large eels	Small eels
0	450	0.24	0.12	0.08
1	195	0.26	0.13	0.11
2	153	0.47	0.22	0.17
3	253	0.46	0.18	0.19
4	208	0.74	0.38	0.28

Table A6. Adjusted average capture rates (per shot), by aquatic macrophyte.

Aquatic macrophyte	No. of shots	Total count	Large eels	Small eels
0	1107	0.37	0.19	0.14
1	85	0.41	0.18	0.14
2	38	0.88	0.21	0.32
3&4	29	0.95	0.31	0.43

Table A7. Adjusted average capture rates (per shot), by snags.

Snags	No. of shots	Total count	Large eels	Small eels
0	459	0.41	0.17	0.16
1	345	0.33	0.18	0.11
2	233	0.51	0.26	0.18
3	109	0.52	0.27	0.17
4	113	0.17	0.05	0.19

Table A8. Adjusted average capture rates (per shot), by undercut bank.

Undercut bank	No. of shots	Total count	Large eels	Small eels
0	701	0.37	0.14	0.17
1	348	0.45	0.28	0.15
2	153	0.50	0.24	0.17
3	57	0.18	0.16	0.03

Table A9. Adjusted average capture rates (per shot), by overhanging vegetation.

Overhanging vegetation	No. of shots	Total count	Large eels	Small eels
0	469	0.49	0.21	0.22
1	549	0.36	0.20	0.12
2	225	0.33	0.15	0.12
3	16	0.17	0.00	0.11

Table A10. Adjusted average capture rates (per shot), by sample.

Sample	No. of shots	Total count	Large eels	Small eels
1	670	0.51	0.26	0.19
2	589	0.28	0.12	0.11
% in first	—	65	68	63

Table A11. Raw and adjusted average capture rates (per shot), by river and site.

River	Site within river	No. shots	Total count		Large eels		Small eels	
			raw	adj.	raw	adj.	raw	adj.
Albert (main)	Chardons Bridge	80	0.01	0.12	0.01	0.06	0.00	0.00
	Darlington Park	40	6.40	1.93	1.63	1.15	4.78	0.78
	Nindooinbah	40	1.33	0.96	0.25	0.22	1.08	0.68
Albert (tribs.)	Canungra Creek	40	0.20	1.39	0.08	0.39	0.13	0.46
	Cedar Creek	40	0.48	0.34	0.13	0.12	0.35	0.23
	Sandy Creek (Albert)	20	0.05	0.04	0.05	0.08	0.00	0.00
Caboolture	Site 1	40	0.15	0.07	0.10	0.11	0.05	0.01
	Site 2	40	0.15	0.13	0.10	0.12	0.05	0.03
	Site 3	40	0.40	0.24	0.28	0.21	0.13	0.06
	Site 4	40	0.70	0.25	0.43	0.21	0.28	0.08
Coomera (main)	Beechmont Rd	40	4.73	1.59	1.25	0.83	3.48	0.75
	Clagiraba Road	43	1.33	0.79	0.19	0.18	1.14	0.43
	Guanaba Creek Road	40	0.23	0.52	0.15	0.19	0.08	0.30
Coomera (tribs.)	Guanaba Creek	40	0.43	0.18	0.08	0.06	0.35	0.12
	Prices Creek	42	0.29	0.13	0.19	0.13	0.10	0.03
	Wongawallan Creek	40	0.28	0.10	0.13	0.06	0.15	0.05
Logan (main)	Cusack Lane	80	0.10	0.17	0.04	0.06	0.06	0.10
	Williams Bridge	20	0.15	0.17	0.15	0.19	0.00	0.00
Logan (tribs.)	Cannon Creek	40	0.10	0.07	0.05	0.08	0.05	0.02
	Sandy Creek (Logan)	40	0.25	0.21	0.13	0.18	0.13	0.06
Mary	Kenilworth	20	0.25	0.24	0.25	0.40	0.00	0.00
	Moy Pocket	80	0.09	0.60	0.09	0.14	0.00	0.00
South Pine (main)	Drapers Crossing	42	0.29	0.22	0.19	0.15	0.10	0.09
	Morrison's Crossing	48	0.67	0.25	0.27	0.17	0.40	0.09
South Pine (tribs.)	Albany Creek	40	0.10	0.19	0.08	0.12	0.03	0.05
	Samford Creek	44	0.27	0.26	0.07	0.12	0.20	0.13
Tinana Creek	Missings	80	0.03	0.16	0.03	0.04	0.00	0.00
	Wilsons Pocket Road	60	0.07	0.05	0.05	0.05	0.02	0.01

Table A12. Adjusted average capture rates (per shot), by fishing pressure.

Pressure	No. of shots	Total count	Large eels	Small eels
Fished	440	0.22	0.16	0.02
Glass-eel harvest	340	0.54	0.25	0.23
Unfished	859	0.42	0.20	0.20

Table A13. Raw average capture rates (per shot), by fishing pressure.

Pressure	No. of shots	Total count	Large eels	Small eels
Fished	440	0.23	0.17	0.07
Glass-eel harvest	340	0.91	0.25	0.66
Unfished	859	0.86	0.26	0.60

LTMP eel surveys

Four rivers (Condamine, Gregory, Mitchell and Warrego) had zero captures, and hence zero variation, and were excluded from the analysis. For the other rivers (totalling 480 shots), table C1 summarises the goodness of fit for the Poisson GLMs. In every model, ‘river’ was a significant contributor after the environmental variables had been screened, indicating that the river differences could not be totally explained by these attributes. The ‘year by river’ interaction was significant

for the total and large eels models, but not in the small eels analysis (but was retained here, for consistency). Tables C2 to C9 list the adjusted means for the significant terms in these models, namely river, river by year, water velocity, snags, voltage, rocks category, level of disturbance, and aquatic macrophyte. The non-significant variables were year; depth, width, water level, riparian continuity and tidality of the river; amperage and gain of electrofishing (correlated with voltage); width of the riparian vegetation; percentage of trees, grasses or no vegetation; canopy cover; overhanging vegetation; type of emergent vegetation; leaf litter; undercut bank; and substrate type. These either had no association with catch rates, or alternately may be correlated with other variables already in the model (and hence had no further statistical contribution).

Table C1. Goodness of fit of the Poisson GLMs for eel captures.

Parameter	Total count	Large eels	Small eels
Residual mean deviance	1.29	1.07	0.70
Deviance explained (%)	35	35	21
Spatial autocorrelation (r)	0.11	0.06	0.12

Table C2. Raw and adjusted (modelled) average capture rates (per shot), by rivers.

River	No. shots	Total count		Large eels		Small eels	
		raw	adjusted	raw	adjusted	raw	adjusted
Daintree	84	2.19	1.64	1.74	1.30	0.45	0.32
Herbert	60	1.63	1.72	1.13	1.23	0.50	0.48
Johnstone	84	1.08	0.82	0.76	0.57	0.32	0.25
Logan	84	0.60	0.77	0.40	0.52	0.19	0.26
Mary	84	0.64	0.75	0.48	0.56	0.17	0.20
Noosa	84	0.16	0.21	0.14	0.18	0.01	0.02

Table C3. Adjusted average capture rates (per shot), for years by rivers.

River	Total count		Large eels		Small eels	
	2000	2001	2000	2001	2000	2001
Daintree	1.98	1.32	1.62	1.00	0.34	0.31
Herbert	1.81	1.64	1.36	1.10	0.43	0.53
Johnstone	0.94	0.71	0.65	0.50	0.29	0.21
Logan	0.66	0.88	0.47	0.57	0.20	0.33
Mary	0.39	1.08	0.28	0.81	0.11	0.28
Noosa	0.17	0.24	0.17	0.19	0.00	0.05

Table C4. Adjusted average capture rates (per shot), by water velocity.

Water velocity	No. of shots	Total count	Large eels	Small eels
High	9	0.48	0.23	0.26
Moderate	388	1.27	0.95	0.23
Low	81	0.90	0.67	0.31

Table C5. Adjusted average capture rates (per shot), by snags.

Snags	No. of shots	Total count	Large eels	Small eels
0	143	0.63	0.41	0.22
1	263	1.05	0.79	0.26
2	52	1.15	0.94	0.22
3	13	1.34	0.96	0.36
4 & 5	9	1.50	1.22	0.28

Table C6. Adjusted average capture rates (per shot), by voltage.

Voltage	Total count	Large eels	Small eels
320	0.69	0.51	0.17
340	0.70	0.52	0.18
500	0.79	0.59	0.20
700	0.92	0.68	0.24
1000	1.15	0.86	0.30

Table C7. Adjusted average capture rates (per shot), by rocks category.

Rocks	No. of shots	Total count	Large eels	Small eels
0	339	0.85	0.63	0.22
1	70	1.11	0.81	0.29
2&3&4	30	1.24	0.98	0.26
5	41	1.33	0.95	0.37

Table C8. Adjusted average capture rates (per shot), by level of disturbance.

Disturbance	No. of shots	Total count	Large eels	Small eels
Extreme	21	1.52	1.22	0.30
High	132	1.08	0.78	0.29
Moderate	101	0.80	0.60	0.21
Low	89	0.98	0.71	0.27
Undisturbed	133	0.84	0.63	0.21

Table C9. Adjusted average capture rates (per shot), by type of aquatic macrophyte.

Aquatic macrophyte	No. of shots	Total count	Large eels	Small eels
0	361	0.90	0.69	0.21
1&2	105	1.06	0.73	0.34
3&4&5	14	1.57	1.10	0.48

Discussion

South-east Queensland eel survey

The low catch rates in the four large dams surveyed suggest that densities of eels are too low in most dams for monitoring of reservoirs by electrofishing to be of much value in assessing trends in eels stocks in the region. Although commercial landings suggest fished dams have been depleted of eel stocks (QFS C fish database), both a fished dam (Wivenhoe) and an unfished dam (Tingalpa) had zero captures, suggesting that factors in addition to fishing could be involved in the low catch rates. One trend identified was lower catches in dense macrophyte beds. In reservoirs in Queensland macrophyte beds can extend out into six metres of water and reach from the bottom to the surface. At this depth efficiency of electrofishing is compromised. Although eels might use these weed beds, the depth of the beds can make detection of stunned eels and netting of observed eels difficult. It is possible that eels were present in the dams at deeper levels, but beyond the range of the electrofisher.

The dam walls themselves may also have affected eel recruitment. The walls of the sampled reservoirs ranged from approximately 10 metres to 50 metres in height. Small eels (<120 mm) can climb vertical surfaces (Jellyman, 1977), and will ascend dam walls if they are damp. Graynoth and Niven (2004 in press) assumed that dams less than 3 m high have little effect on movements of

New Zealand longfinned eels, but flows over low weirs are much more likely to be sustained in New Zealand than they are in South-east Queensland. In New Zealand high dams are known to exclude or severely reduce recruitment of eels. For example no eels were found in Lake Mahinerangi, a New Zealand hydro-electric storage (Allibone, 1999).

Elvers and glass eels leaving the water to bypass weirs are more at risk of predation. Opportunities for elvers and glass eels to migrate up dam walls are limited to those times when water is spilling over the dam, creating damp surfaces, and possibly also to rainfall events which create damp surfaces. Failure to migrate upstream and mortality from predation can be expected to increase with increasing dam height. Higher dams also tend to spill less often. Thus replenishment of dam stocks following fishing activities or loss of downstream migrants can be compromised. Lenthalls dam spills in most years during summer and this may explain the better catches from this dam compared to the others sampled. No small eels (<300mm) were caught in any of the dams suggesting recruitment must have been poor in recent years or that small eels were not using the lacustrine habitats sampled by electrofishing.

A number of environmental parameters were found to have a significant influence on river and stream catches of eels. The first of these is depth. The depth effect is most pronounced for small eels, with catches tending to be greater in shallow waters. Jellyman *et al.*, (2003) also noted that juveniles of both species of New Zealand eels preferred shallow water. Broad *et al.* (2001a) found a similar result for *A. dieffenbachia*, and reported that eels were significantly longer in pools than in riffles. In our study shallow waters generally corresponded to riffle habitats, cascades and backwaters, where we observed large concentrations of juvenile eels (see Table A5). Use of these shallow areas may have afforded small eels some protection from predation by large eels and other predatory fishes. The depth effect was less marked for large eels, but there is a slight tendency for large eels to be more prevalent in deeper water. This includes deeper holes at the bottom of cascades.

Other habitat variables with significant positive effects on catch rates of both large and small eels were rocks and aquatic macrophytes. We conclude that eels take advantage of the cover provided by rocks and macrophytes in rivers and streams. The macrophyte result contrasts with that of dams, but macrophyte beds in rivers and streams were shallower and easier to sample by electrofishing than those in dams. Jellyman *et al.* (2003) noted that large (>500mm) eels in New Zealand were strongly associated with cover, which included undercut banks, weed and in-stream debris.

In our survey snags or large woody debris also affected eel captures. This was more obvious for large eels, with higher catch rates coming from moderate densities of snags than from low densities or high densities. Small eels were more uniformly distributed. Similarly, the data suggest that large eels were more abundant in sites with moderate levels of undercut banks than in sites without undercut banks or with extensive undercutting. In our surveys, small eels were rarely captured in sites with extensive undercuts. We would have anticipated increasing abundance of large eels with extent of undercut banks. The drop off in catch rates at the most extensive undercut bank category was unexpected, but might be explained by greater potential to avoid capture in such situations, especially at a 5 metre level of sampling. If the entire bank in the shot zone is undercut, then there is more potential for an eel to remain undetected if it moves laterally, than in situations where the undercuts are discontinuous. Extensive snags may have a similar effect. The very low capture rates of small eels in such situations may in part be explained by failure to observe them but probably also by exclusion by larger eels or other large predatory fish which use undercuts.

In contrast, overhanging vegetation was negatively associated with catch rates of both small and large eels. This may in part be related to eels being more visible in areas without overhanging vegetation and more difficult to dip-net from under overhanging vegetation, but there does appear to be a real effect. It is generally more disturbed areas that lack overhanging vegetation. These same areas may also favour macrophyte growth, with which eels have been positively associated in this study. Macrophyte abundance was weakly negatively correlated with overhanging vegetation.

The SEQ eel surveys were one-off samples, so no comparison could be made between years. However, between-site differences were detected which could not be explained by the evaluated environmental parameters alone. If between-site differences can be detected, then the data should be sensitive enough to pick up a signal if significant changes in density occur through time.

It is interesting that no significant difference in eel catch rates was detected between sites in fished and unfished catchments, although the variability within both fished and unfished rivers, and therefore the power of this type of test, was such that even a large difference would have been difficult to detect. The large difference observed (lower in fished catchments) was not statistically significant. Fishing impacts may have been minimal compared to other effects in the rivers, since commercial fishing is restricted to weir pools within the fished rivers. Tributary streams and unimpounded sections remain unfished. Movement of eels between fished and unfished areas of the rivers is likely, and this could mask any local impacts. The proportion of the sampled rivers that is fished is probably not large enough to significantly affect the river system as a whole. Glass eel harvest was not observed to affect abundance of large or small eels, though again the power of this test was low. Current levels of glass eel harvest may be below those where density dependent impacts on recruitment might occur. Enough glass eels may be escaping capture for density independent recruitment to still take place.

LTMP eel survey data

The results from this data are largely consistent with the results of the SEQ eel survey. A key difference between the two sets of data is that habitat variables in the SEQ eel survey were collected at a 5 m shot level, whilst the LTMP data is collected at a 50 m shot level. The other key difference is that the LTMP program data comes entirely from the main stream of key rivers, whereas the SEQ eel surveys also included small tributary stream sites. The SEQ surveys used both backpack and boat mounted electrofishing, whereas the LTMP data is restricted to boat mounted electrofishing and areas deep enough for boating. Therefore it is not surprising that depth was a significant variable in the SEQ eel survey data, but not significant in the LTMP data, as the range of depths sampled by the latter method were more restricted.

Snags, rocks and aquatic macrophytes were all found to be significant influences on LTMP eel catch rates. The same variables were identified as significant in the SEQ eel survey data. All three variables were positively associated with LTMP total eel catch rates. In the case of small eels the effect of snags was less than that for large eels, and adjusted mean captures of small eels were lower in the most abundant snag categories (4&5) than in the preceding moderate (3) category. A similar result was noted with the SEQ eel survey data. In contrast to this result, Broad *et al.* (2001b) found the New Zealand species of longfinned eel *A. dieffenbachii* to be negatively associated with wood debris in the Taieri River. For modelled LTMP data, adjusted mean recaptures of large eels increased with abundance of snag cover and captures of both large and small eels increased with increasing rocks and macrophytes. This is also consistent with SEQ eel survey results.

Disturbance was also found to influence catch rates of eels, with adjusted catch rates higher at more disturbed sites. This agrees with the SEQ eel survey results, where catch rates had a negative

relationship with overhanging vegetation. A number of the variables not selected by the GLM as significant for the LTMP eel data may have been correlated with the index of disturbance, for example, riparian continuity and width and overhanging vegetation. Similarly in New Zealand, Broad *et al.* (2001b) found *A. dieffenbachii* to be more abundant in catchments with pasture, than in catchments with native forest. That eels are favoured by disturbance is an interesting result. One of us (M.H.) has observed *A. reinhardtii* in urban drains where no other native fish survive. Thus eels appear to be more tolerant of poor water quality and disturbed habitats than most other native fish species. Reduced competition with, and predation by other fish species in disturbed areas may favour eels. Eel catch rates were also positively associated with voltage settings used during electrofishing. Voltages were set higher by operators in low conductivity conditions and were set lower in higher conductivity conditions in an effort to standardise power outputs. The lowest conductivities were in the northern rivers and these rivers tended to have higher catch rates of eels. Whether voltage was a real effect on catch rates or an artefact of where sampling took place is open to conjecture.

The most important result with the LTMP data is that in every model, 'river' was a significant contributor after the environmental variables had been screened, and the 'year by river' interaction was significant for the total and large eels models. This suggests that the LTMP eel data should be suitable for monitoring trends in stocks of large eels, between rivers and between years. The LTMP sampling is mainly confined to river pools and does not sample shallower habitats that tend to be favoured by smaller eels (see SEQ eel survey data). This is possibly why these data were unable to detect between-river or between-year differences in small eel numbers. Nevertheless large eel numbers give some indication of the abundance of potential spawners and this is an important consideration when managing eel stocks.

The LTMP eels surveys have been set up to monitor a range of freshwater fish stocks in Queensland. These surveys are planned to continue into the foreseeable future. The analysis of the LTMP data suggests that these surveys are suitable for detecting changes in abundance of large eels (i.e. those potentially available to the adult eel fishery) and total eel numbers. Therefore LTMP surveys can be used to generate an index of abundance for legal sized longfinned eel stocks. Should downward trends be detected in electrofishing CPUE over several years, then such a decline could be used as an early warning signal. This could lead to activation of surveys along the lines of the SEQ eel surveys. The SEQ eel surveys covered a wider range of habitats and were at a finer scale and may be more useful in confirming recruitment of smaller eels. These studies may also help confirm whether declining large eel stocks are related to reduced recruitment of small eels or other factors that are impacting on larger eel size classes.

Identification of habitat factors significantly influencing catch rates in both the LTMP and the SEQ eel surveys is interesting. One of the main reasons for identifying such habitat factors was to remove sampling variation and so increase the power to detect abundance changes. Since these factors explain between 21 per cent and 73 per cent of the deviance, this approach appears to have been successful. Moreover, identification of habitat factors raises the possibility of developing a GIS that could predict areas most likely to provide suitable eel habitat or contain high densities of eels. Broad *et al.* (2001b) developed models based on biotic and physicochemical variables to predict the probability of occurrence of *A. dieffenbachii* in the Taieri River catchment New Zealand. At a macro scale, occurrence of eels declined with increasing elevation and for a given elevation was higher in tussock and pasture catchments and lower in pine and native forest catchments. A map was generated from this macro-scale model and showed areas of high, medium and low probability of occurrence of longfinned eels.

Hoyle and Jellyman (2002) suggested New Zealand longfinned eels need catchment reserves to provide for sufficient spawner escapement. Graynoth and Niven (2004 in press) developed a GIS system to estimate the proportion of longfinned eel biomass in New Zealand's West Coast and Southland that exists in areas not currently fished. This macro-scale GIS habitat based model uses mean annual flow and stream reach gradient as the key predictors of biomass distribution. The macro-scale habitat variables were selected in preference to micro-habitat variables such as water depths, velocities and substrates, as such detailed information was not available at a large geographical scale. Biomass was preferred over abundance indices as the density of large eels (*i.e.* eels >400 mm TL) was strongly correlated with biomass. Overall densities, on the other hand, were strongly influenced by the number of small eels. The number of large eels (potential spawners) is more important when considering adequacy of reserves, thus predictors of biomass were useful for sustainable management of the fishery. There was also some evidence that biomass of small eels may increase following removal of large eels, due to improved density dependent survival. Thus biomass was a useful predictor of the potential of eel reserve locations.

The current habitat data collected for both the LTMP and the SEQ eel surveys has been at more of a micro-habitat level. For some of these (e.g. riffle, pool, backwater) it may be possible to estimate parameters remotely. It may also be possible to estimate density of large woody debris from air photos etc in the main stream of river systems. However macro-scale variables such as stream reach gradient, mean annual flow, elevation and stream order are more easily obtainable for entry into a broad regional GIS model. Whether these macro-variables will be as useful in an Australian context as in New Zealand as predictors of eel biomass remains unknown. Their potential should be investigated.

If they are shown to have strong predictive capacity for either eel biomass or abundance of different size classes, then they could be used to develop a GIS model to assist with selection of eel catchment reserves in eastern Australia.

The micro-habitat variables identified as important from the SEQ eel surveys and the LTMP could perhaps be incorporated into catchment or sub-catchment GIS models for eel stocks, although collection of such data would be difficult for GIS models covering larger geographical areas. Nevertheless the knowledge of important microhabitats gained from this current analysis could be useful information for rehabilitation or wetland creation works, where the aim is to favour either recruitment of small eels, supporting stocks of large eels or both. Certainly there is potential for such programs under a number of recent Commonwealth Government natural resource management funding initiatives.

Conclusions

1. Current densities of eels in large impoundments are too low for these locations to be suitable for long-term monitoring of eel stocks.
2. The existing LTMP for river eel stocks appears suitable for use as an index of abundance for large eels and total eel catch. Numbers of large eels are critical in terms of future potential spawners, and any downward trends in abundance could be used as an early warning for future spawner production.
3. There is no statistically significant evidence for a decline in eels in SEQ rivers that is attributable to either adult eel harvest or glass eel harvest. Nevertheless there was a non-significant trend for eel catch rates to be lower from sites in fished catchments.
4. A number of habitat variables have been identified as influencing catch rates of both small and large eels in riverine habitats. This knowledge provides an option to manipulate, create or rehabilitate habitats to favour large eels, recruitment of small eels, or both should such actions be deemed necessary or desirable.

5. If the LTMP detects declines in large eel or total eel abundance within a single river or across several rivers, then a SEQ style survey could be re-implemented to examine a broader range of habitat types (including those favoured by smaller eels) to better define at which stage of the eel life history losses are occurring.
6. The value of macro-habitat variables (e.g. mean annual flows, stream reach gradient) for predicting the distribution of eel biomass in Australian catchments needs to be investigated. If these variables prove to be useful predictors, then they could be incorporated into a GIS system that could be developed for managing the eel fishery through appropriate distribution
7. of eel fishing reserves (see Chapter 6).

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Chapter 2: Mark recapture experiment

Simon Hoyle

Abstract

The primary objectives of this mark-recapture experiment, as a component of the overall eel project, were to determine survival and growth rates for longfinned eels, and to estimate population densities in support of the age structure sampling. However, results of this experiment combined with results of the tank experiment suggest that only the objective related to growth can be met.

Survival rates estimated in this experiment are considerably below those estimated from the age structure sampling. In many cases they suggest survival rates less than 50 per cent per year. This is not surprising in stream-based plots where emigration is likely to occur, but low survival rates were also estimated in the three dam-based sites. Future radio-tracking studies may provide more information on emigration and mortality rates.

Growth rate varied substantially between locations, a result similar to virtually all analyses of eel growth rates. Eel growth is in general highly variable between individuals, locations, and time periods. Seasonal variation in eel growth rate in the wild was moderately supported by the model, a result that tallies with the results of the tank experiment, and lends further support to our ageing validation results.

Objective

Estimate population parameters required for a management model. These include survival, density, age structure, growth, age and size at maturity and at recruitment to the adult eel fishery. Estimate their variability among individuals in a range of habitats.

Introduction

A mark-recapture experiment is an efficient way of obtaining a great deal of demographic and behavioural information. Information is obtained at an individual level, and different assumptions are involved from those that apply to data obtained at a population level. In the context of this study such experiments are therefore useful to validate, calibrate, and extend results from the population sampling. Mark-recapture models are a rapidly developing area of statistics, and their power and flexibility have increased significantly in recent years, with many papers being published each year. Given sufficient data and the right design, parameters that can be estimated include recruitment, survival, growth, immigration, emigration, and catchability rates, as well as population size and density. Their power is increased by conducting multiple experiments and analysing the results jointly.

An electrofishing and trapping-based mark-recapture experiment was used to estimate survival rates, catchability, and growth rates of adults in unfished waterways. Growth is perhaps the most important parameter determining the relative effect of fishing pressure on numbers of eels reaching maturity. Mortality estimated by mark recapture is actually a composite of mortality and emigration, but can be informative about ecology and the factors influencing population dynamics (Hoyle, Pople *et al.*, 2001). Catchability is important for translating catch rates into absolute numbers and biomass estimates. This experiment also has the potential to estimate population density, which can then be used to calibrate CPUE from electrofishing or fyke netting, and to improve the CPUE model.

However, a number of factors complicate mark-recapture analyses, including variation in catchability between individuals and times, and between individuals that have been caught previously and those that have not. The effectiveness of electrofishing declines with the size of fish (Sullivan 1956; Junge and Libosvarsky, 1965, Reynolds and Simpson 1987). Furthermore, people dipnetting stunned fish are more likely to overlook smaller specimens than large ones (Reynolds, 1985). Given this change in effectiveness with size, population estimates using electrofishing require either grouping by size (Sullivan 1956), use of size as a covariate, or use of a model that is robust to varying catchability by size.

However, combinations of data from multiple gears improve estimates of population size from closed mark-recapture experiments by reducing the catchability variation between individuals. Electrofishing can result in significant mortality in salmonids (Hughes, 1998), a wide range of sublethal injuries (Hauck 1949; Sharber and Carothers, 1988; Mesa and Shreck, 1989; Holmes *et al.*, 1990; Hollender and Carline 1994; Thompson *et al.* 1997a), and a reduction in growth rate (Gatz *et al.*, 1986; Dwyer and White, 1995; Thompson *et al.*, 1997b). These problems can be reduced by minimising application of power.

Methods

The waterways were largely closed in order to minimise loss of eels due to movement. Larger, older eels are generally highly territorial (Brad Pusey, pers. comm.) so use of physically closed waterways was not essential. We used lagoons adjacent to river systems, and river pools that were fairly discrete entities. The location of sites is shown in Figure 1.

The experiment was carried out over a period of two years with six sites sampled twice each year. These sites were not fished commercially during the experiment. We used Pollock's robust design (Nichols *et al.*, 1984; Pollock 1982), which involves both a short-term (closed) and a long-term (open) mark-recapture experiment.

Before each two-day session, at each site stopper nets were set to block ends of a reach where closure was not already complete by natural means such as rock bars. Sites were classified according to a number of habitat variables, using the system described for the age structure and CPUE sampling.

Fyke nets and baited traps were set on the first evening at the site. The following morning the traps were cleared and all eels tagged and released. Then the site was electrofished using the standard methodology and captured eels marked and released. All recaptures were recorded. After completion of electrofishing fyke nets and traps were reset and cleared the next morning, followed by a second electrofishing session. Stopper nets were removed after this electrofishing session. This gave a four-stage closed mark-recapture experiment in two days of sampling per site. The entire process was repeated at six month intervals the open mark-recapture experiment.

All eels captured were immediately anaesthetised with clove oil, measured, uniquely tagged, injected with oxytetracycline, and released within 20 metres of the capture site after they recovered from anaesthesia. Eels over 25 cm were tagged with internal PIT tags. These tags have a far higher retention rate in eels than external tags, which also promote infection and reduce survival rates. This is significant for an animal with a high survival rate as expected for eels. Eels smaller than 25 cm were injected with visual implant elastomer in three of eight positions, in a unique combination of three colours. Oxytetracycline was administered by intraperitoneal injection at a rate of 75 milligrams per kilogram of body weight (Chisnall and Kalish, 1993).

During each trapping session traps were distributed randomly around the site. The 20 large eel traps were placed approximately three metres from the bank, and the 20 minnow traps were placed

approximately 1 metre from the bank. Each site was divided into four quadrants, with five large traps and five small traps allocated to each quadrant. Within each quadrant 10 random numbers between 1 and 50 were generated and each trap was placed at the point (distance = rand x quadrant length/50), where distance is the distance from one end of the quadrant. Fyke nets were placed at each end of the site next to the bank, in order to catch eels swimming parallel to the bank. On the second night's trapping both fyke nets were placed on the opposite bank, if this was possible. Large traps and fyke nets were baited with two pilchards and minnow traps with one pilchard.

The standard methodology for carrying out electrofishing used a 7.5 kilowatt electrofishing unit in a 5.6 metre aluminium boat. The entire length of bank was sampled during each session. Before each session we used a Horiba meter to measure temperature at the surface and on the bottom, and water conductivity. Turbidity was measured using a Secchi disk.

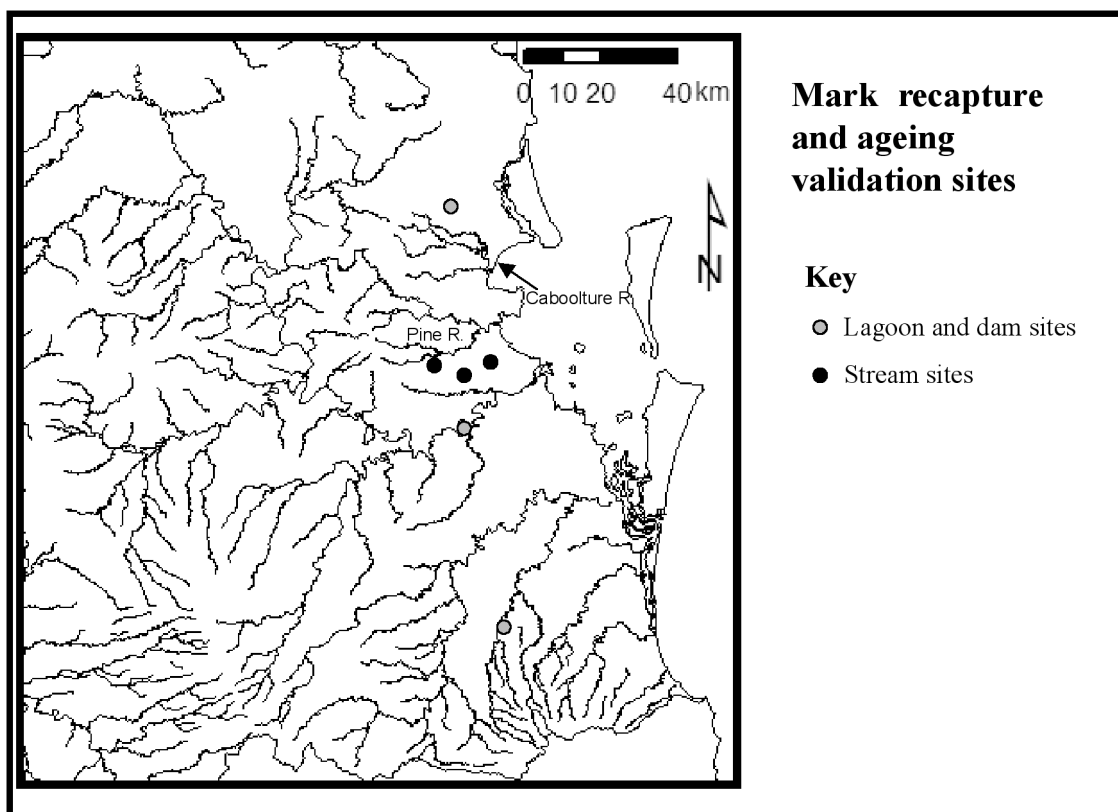


Figure 1: Mark recapture and ageing validation sites.

Analysis of the mark-recapture experiment was carried out using the programs CAPTURE (Otis, Burnham *et al.* 1978) and MARK (White and Burnham, 1999). CAPTURE was used for closed population estimates (i.e. short period analyses, assuming no death or emigration). CAPTURE uses eight mark-recapture models, each of which makes different assumptions about variation in catchability between individuals and capture sessions. Model *Mh* assumes probability of capture varies between individuals. Model *Mb* assumes different probability of capture for individuals caught previously and those never caught. Model *Mt* assumes that probability of capture changes between capture sessions. These models can be combined (i.e. model *Mth* includes time-dependent and individual variation in capture probability). Model *Mo* assumes the same catchability for all individuals and between capture sessions.

Program CAPTURE contains statistical tests to compare the fit of different models. However, these models are not very powerful, and it is often better to choose models *a priori*, based on knowledge of the study animal's behaviour and the conduct of the experiment.

Program MARK was used for open population analyses of survival and migration between the six-monthly sessions. For each location, combinations of the following models were examined: constant survival S_c , time-varying survival S_t , and survival affected by capture S_f ; constant catchability P_c , and time-varying catchability P_t . Goodness of fit was examined using the tests in the program RELEASE (Burnham *et al.* 1987) and the bootstrap goodness of fit tests in MARK. Models were adjusted for overdispersion using the \hat{c} parameter, calculated by dividing the deviance of the model $S_t P_t$ by the number of parameters. Models were compared using the Akaike Information Criterion, corrected for overdispersion and number of parameters (QAICc, Burnham and Anderson 1998). Relative likelihoods were calculated as the QAICc weight of the model of interest divided by the QAICc weight of the best model in the set.

Tag recapture data were analysed for growth using a modification of the Francis (1988) method (GROTAG), using maximum likelihood to estimate growth parameters and variability. This method permits estimation of growth rate g , coefficient of variation of growth rate v , measurement error m , and outlier contamination p . Growth rate was assumed to be linear rather than following a von Bertalanffy growth curve, so a single parameter g_s was estimated for each site s . Parameters were also estimated for the effects of tagging (rt) and electrofishing (re) on growth. The form of the growth equation was $\Delta L = g_s \Delta t + re + rt + \varepsilon$, where the error term $\varepsilon = g_s \Delta t$, proportional to expected growth without electrofishing and tagging effects. Modelling began with the simple model $\Delta L = g \Delta t + \varepsilon$, with more parameters added subject to tests of model support, using the Akaike information criterion. An Excel spreadsheet model was used (see Haddon 2001, p 210 for Excel implementation of the model described by Francis (1988)).

Results

For the closed capture sessions there were prior expectations about the appropriate model to apply to estimate population size. These expectations are outlined below, and were also examined using the model selection algorithms in the program CAPTURE. A concern was that there would be temporal variability, since electrofishing and trapping have different catchabilities both by size and overall. It was also known anecdotally that eel trappability tends to vary with time. It was also expected that eels would vary in catchability by size, since this was observed in the electrofishing age sampling, and in previous studies (Rossi *et al.*, 1987) (Naismith and Knights, 1990). There was also concern that there might be some behavioural response to the capture processes. Eels enter traps by choice, and might learn to avoid them after the experience of being tagged. Eels may also learn to evade capture by electrofishing by hiding in deep holes which cannot be fished effectively.

The number of animals caught at each location in each capture session and on each occasion are reported in Table 1 to Table 6, together with the cumulative numbers caught within a session, numbers newly caught on each occasion and frequencies of capture. The closed capture sessions carried out using traps in Jess dam (Table 1) and the Sherwood arboretum dam (Table 5) mostly resulted in low to zero short-term recapture rates. This implied either large population sizes or a behavioural effect of capture on recapture probability. Possible behavioural effects include either short or long-term trap avoidance, emigration (streams only), and capture-induced mortality. Model selection results overall for all locations (Table 7) supported models $Mtbh$, Mo , Mth , Mbh , Mtb , Mt and Mh in decreasing priority. However, Model Mtb was not available in program CAPTURE, and Mbh rarely gave acceptable results.

Table 1: Capture information for Jess Dam.

Location	Session	Occasion	Animals caught	Total caught	Newly caught	Frequencies
Jess dam		$j=$	$n(j)=$	$M(j)=$	$u(j)=$	$f(j)=$
	1	1	13	0	13	28
		2	5	13	5	0
		3	7	18	7	0
		4	3	25	3	0
				28		
	2	1	15	0	15	40
		2	3	15	3	2
		3	26	18	24	0
				42		
	3	1	14	0	14	26
		2	14	14	13	1
				27		
	4	1	33	0	33	28
		2	5	33	0	5
				33		

Table 2: Capture information for Kedron Brook at Grange.

Location	Session	Occasion	Animals caught	Total caught	Newly caught	Frequencies
Kedron Brook — Grange		$j=$	$n(j)=$	$M(j)=$	$u(j)=$	$f(j)=$
	1	1	17	0	17	29
		2	26	17	19	7
				36		
	2	1	17	0	17	18
		2	1	17	1	5
		3	10	18	5	0
				23		
	3	1	5	0	5	15
		2	10	5	10	0
				15		
	4	1	12	0	12	14
		2	12	12	7	5
				19		

Table 3: Capture information for Kedron Brook at Keperra.

Location	Session	Occasion	Animals caught	Total caught	Newly caught	Frequencies
Kedron Brook — Keperra		$j=$	$n(j)=$	$M(j)=$	$u(j)=$	$f(j)=$
	1	1	1	0	1	13
		2	8	1	8	4
		3	2	9	2	0
		4	8	11	4	0
		5	2	15	2	0
				17		
	2	1	3	0	3	14
		2	9	3	9	3
		3	1	12	1	0
		4	7	13	4	0
				17		
	3	1	3	0	3	12
		2	4	3	4	1
		3	5	7	5	0
		4	2	12	1	0
				13		
	4	1	7	0	7	8
		2	2	7	2	3
		3	5	9	2	0
				11		

Table 4: Capture information for Samford.

Location	Session	Occasion	Animals caught	Total caught	Newly caught	Frequencies
Samford		$j=$	$n(j)=$	$M(j)=$	$u(j)=$	$f(j)=$
	1	1	1	0	1	19
		2	15	1	14	6
		3	18	15	11	1
				26		
	2	1	6	0	6	14
		2	9	6	8	2
		3	2	14	2	1
		4	4	16	1	0
				17		
	3	1	1	0	1	5
		2	4	1	4	0
				5		
	4	1	7	0	7	9
		2	1	7	0	2
		3	5	7	4	0
				11		

Table 5: Capture information for Sherwood arboretum.

Location	Session	Occasion	Animals caught	Total caught	Newly caught	Frequencies
Sherwood		j=	n(j)=	M(j)=	u(j)=	f(j)=
	1	1	25	0	25	31
		2	6	25	6	0
				31		
	2	1	4	0	4	17
		2	13	4	13	0
				17		
	3	1	3	0	3	4
		2	1	3	1	1
		3	2	4	1	0
				5		

Table 6: Capture information for Tabragalba lagoon.

Location	Session	Occasion	Animals caught	Total caught	Newly caught	Frequencies
Tabragalba		j=	n(j)=	M(j)=	u(j)=	f(j)=
	1	1	5	0	5	118
		2	44	5	43	21
		3	5	48	5	1
		4	27	53	27	0
		5	82	80	60	0
				140		
	2	1	34	0	34	67
		2	41	34	37	4
				71		
	3	1	20	0	20	37
		2	21	20	19	2
				39		
	4	1	15	0	15	16
		2	1	15	1	1
		3	2	16	1	0
				17		

Table 7: Model selection weights for alternative closed mark-recapture models, as calculated by program CAPTURE. The various models are described in the text.

Location		M(o)	M(h)	M(b)	M(bh)	M(t)	M(th)	M(tb)	M(tbh)
Jess	1	NA							
	2	0.73	0.72	0.03	0	0.73	0.7	0.53	1
	3	0.68	0.03	0.26	0.84	0.53	1	0	0.85
	4	0	0.34	1	0.1	0.5	0.07	0.83	0.12
Grange	1	0.83	0.16	0	0.84	0.42	0.18	0.1	1
	2	0.13	0	0.52	0.27	0.31	0.56	1	0.4
	3	NA							
	4	0.68	0.03	0.26	0.84	0.53	1	0	0.85
Keperra	1	0.37	0.18	0	0.17	1	0.79	0.26	0.28
	2	0.44	0.19	0	0.28	1	0.97	0.26	0.42
	3	1	0.83	0.42	0.74	0	0.45	0.35	0.78
	4	0.85	0.73	0.51	0.9	0	0.76	0.63	1
Samford	1	0.74	0.79	0.02	0	0.32	0.54	0.44	1
	2	0.87	0.73	0.67	0.92	0	0.54	0.81	1
	3	NA							
	4	0.78	0.62	0.24	0.55	0	0.59	0.27	1
Sherwood	1	NA							
	2	NA							
	3	1	0.84	0.6	0.96	0	0.61	0.45	0.98
Tabragalba	1	0.13	0	0.07	0.08	1	0.68	0.34	0.21
	2	0.84	0.15	0	0.85	0.41	0.18	0.09	1
	3	0.85	0.16	0	0.86	0.41	0.18	0.08	1
	4	0.17	0.07	1	0.45	0	0.33	0.74	0.43
Average		0.616	0.365	0.311	0.536	0.398	0.563	0.399	0.740

Population size estimates for all methods shown tended to be similar within an occasion, with wide confidence intervals. The models incorporating behavioural effects, which were largely not reported, were lower in the cases when they were reported. This is expected since trap avoidance biases population estimates upwards. Four locations showed a downward trend in estimated abundance, while for Sherwood arboretum only one estimate could be calculated, and at Keperra estimates varied widely. At Keperra the final estimate was the lowest observed there. These downward trends could have been contributed to by the effects of capture causing either mortality or long-term trap avoidance. Tabragalba lagoon experienced a severe algal bloom in summer months, associated with a drought, which may have caused significant eel mortality. Drought conditions prevalent during the study may also have contributed to mortality or emigration from other sites.

Table 8: Population size estimates (N) under three models for each location by mark-recapture session. Lower (L) and upper (U) 95 per cent confidence limits are given, based on profile likelihood if available and normal approximation if not.

Location	Session	Start Date	Assume Mo			Assume Mth			Assume Mh		
			N	L	U	N	L	U	N	L	U
Jess dam	1	10/6/1999	NA			NA			NA		
	2	4/4/2000	242	111	1827	366	122	1345	442	146	1578
	3	10/3/2000	149	52	618	NA			365	87	1911
	4	1/30/2000	77	42	161	NA			111	59	264
Grange	1	12/20/1999	63	45	113	NA			96	58	199
	2	5/8/2000	46	28	108	45	27	129	55	32	120
	3	10/23/2000	NA			NA			NA		
	4	2/15/2001	27	19	54	NA			38	24	89
Keperra	1	7/14/1999	37	20	103	35	22	86	38	22	98
	2	4/18/2000	47	21	152	42	23	118	49	25	147
	3	10/17/2000	64	19	1036	68	22	344	85	24	453
	4	2/1/2001	18	11	55	17	11	49	21	13	63
Samford	1	7/29/1999	42	29	76	65	34	211	56	35	120
	2	3/21/2000	35	20	96	72	27	300	66	27	238
	3	10/12/2000	NA			NA			NA		
	4	2/8/2001	24	13	72	24	12	102	31	14	114
Sherwood arboretum	1	5/5/2000	NA			NA			NA		
	2	10/10/2000	NA			NA			NA		
	3	1/18/2001	9	5	112	10	5	51	13	5	69
Tabragalba	1	7/7/1999	426	312	624	471	321	474	471	329	719
	2	4/6/2000	344	170	1021	NA			632	273	1628
	3	10/5/2000	200	79	1103	NA			381	127	1363
	4	2/6/2001	84	28	1599	214	43	1496	145	38	767

Open population analyses

Survival rate (less emigration) over a six month period and probability of capture were estimated with several different models (Table 8), and the fit of those models compared using the Akaike information criterion (Table 9). Models with relative likelihood greater than 0.2 are reported in Table 9. The models with different survival following first capture were well supported, and survival after first capture was usually estimated as lower than subsequent survival, though none of the differences were statistically significant. Survival here is confounded with emigration and with permanent trap avoidance.

Goodness of fit tests using program RELEASE were inconclusive in all cases due to low sample sizes. Bootstrap goodness of fit tests were more successful, and did not show lack of fit for any of the models. These tests have been shown to be biased low (White 2002), with the bias increasing as the number of occasions increases and the apparent survival rate increases. In this case both the number of occasions and apparent survival were low.

Table 9: Survival rate and catchability estimates for each location under plausible models. Models for survival (S) and probability of capture (P) are either constant (c), different for first and subsequent captures (f), or varying with time (t).

Model	Location		Survival (6 months)	SE	L	U	Capture	SE	L	U	Relative likelihood
Sc Pc	Jess		0.726	0.356	0.073	0.989	0.073	0.054	0.016	0.275	1
	Grange		0.588	0.123	0.346	0.794	0.371	0.107	0.194	0.591	1
	Keperra		1	1E-07	1	1	0.188	0.042	0.119	0.285	1
	Samford		0.750	0.142	0.405	0.930	0.309	0.115	0.134	0.562	1
	Sherwood		0.365	0.251	0.064	0.827	0.378	0.253	0.069	0.833	0.938
Sf Pc	Jess	S 1st	0.151	0.098	0.038	0.441	0.275	0.167	0.069	0.661	0.876
		S other	1	0	1	1					
	Grange	S 1st	0.376	0.121	0.181	0.623	0.455	0.119	0.245	0.681	0.725
		S other	0.770	0.176	0.322	0.959					
	Keperra	S 1st	0.985	0.323	0	1	0.192	0.088	0.072	0.420	0.319
		S other	1	0	1	1					
	Samford	S 1st	0.707	0.207	0.254	0.945	0.320	0.124	0.134	0.589	0.335
		S other	0.790	0.216	0.227	0.980					
	Sherwood	S 1st	0.659	0.765	0.002	0.999	0.271	0.260	0.028	0.830	0.486
		S other	0.139	0.187	0.007	0.776					
St Pc	Sherwood	S1	0.451	0.340	0.053	0.924	0.452	0.312	0.065	0.907	1
		S2	0.087	0.104	0.007	0.553					
	Tabragalba	S1	0.696	0.099	0.477	0.852	0.428	0.094	0.261	0.613	0.883
		S2	0.336	0.076	0.206	0.497					
		S3	0.249	0.119	0.087	0.536					
St Pt	Tabragalba	1	0.786	0.142	0.412	0.951	0.338	0.102	0.173	0.555	1
		2	0.212	0.074	0.102	0.390	0.661	0.181	0.286	0.905	
		3	0.499	149.578	0	1	0.329	67.122	0	1	

Growth analysis

A total of 115 eels, tagged and recaptured in different sessions, were used in this analysis. Time at liberty ranged from 5 to 83 weeks, and size at tagging for eels used in the analysis ranged from 210 to 970 mm. Sizes within this range were roughly normally distributed, with average length at tagging 526 mm. Analyses showed that a model with 12 of the 13 parameters was best supported by the data (Table 10). There was little support for an effect of electrofishing on growth, though this effect was largely confounded with site. There was moderate support for seasonality in growth, with peak growth estimated at $w = 0.128$, equivalent to mid-February. Mean annual growth rate was estimated at 86 mm for Jess Dam, 35 mm for Grange, 24 mm for Keperra, 27 mm for Samford, 62 mm for Sherwood, and 1 mm for Tabragalba.

Table 10: Results of GROTAG analysis

Parameter	Basic (g, nu)	+ sites (g _s)	+ msr error (s)	+ ssn (u, w)	+ outl (p)	+ tag (rt)	+ elect (re)	- elect - ssn
Lik	556.94	529.75	483.62	481.80	473.34	465.77	465.75	467.85
np	2	7	8	10	11	12	13	10
AIC	1117.88	1073.50	983.23	983.59	968.69	955.54	957.50	955.71
nu	5.221	10.831	0.792	0.965	1.066	0.967	0.986	0.978
g	10.35	11.67	91.41	76.47	70.33	85.95	84.31	93.69
u	0	0	0	0.594	0.523	0.497	0.487	0.000
w	0	0	0	0.120	0.080	0.128	0.122	0.000
s	0	0	12.566	11.093	7.453	6.255	6.270	6.293
p	0	0	0	0	0.024	0.023	0.024	0.025
rt	0	0	0	0	0	-5.488	-5.509	-5.507
re	0	0	0	0	0	0	0.747	
g (Grange)		-6.402	-70.813	-53.284	-41.382	-51.100	-50.110	-49.031
g (Keperra)		-9.320	-76.532	-61.727	-53.789	-62.042	-61.216	-73.204
g (Samford)		-8.339	-71.126	-56.677	-50.259	-58.643	-58.103	-65.657
g (Sherwood)		-4.527	-52.476	-24.260	-18.283	-23.751	-23.093	-47.326
g (Tabrag)		-8.512	-94.492	-79.667	-74.001	-85.188	-83.570	-92.909

Discussion

The primary objectives of this mark-recapture experiment, as a component of the overall eel project, were to determine survival and growth rates for longfin eels, and to estimate population densities in support of the age structure sampling. However, results of this study combined with results of the tank experiment suggest that only the objective related to growth can be met.

Survival rates estimated in this experiment are considerably below those estimated from the age structure sampling. In many cases they suggest survival rates less than 50 per cent per year. This is not surprising in stream-based plots where emigration is likely to occur, but low survival rates were also estimated in the three dam-based sites. Eels can also leave dam-based sites by migrating overland during wet weather. It is possible that drought conditions experienced during the course of this research may have led to greater emigration rates or cannibalism among eels. It is also possible that eels learned to avoid traps or electrofishing. These survival estimates are therefore not useful for supporting age structure sampling results in developing models of eel population dynamics. We suggest radio-telemetry studies of eels may be useful for estimating emigration rates and mortality.

The second objective of this mark-recapture experiment, to estimate eel growth rates from mark-recapture in the wild, may have been somewhat biased by stunting due to electrofishing, in those eels captured using this technique. This problem was indicated by the tank experiment carried out during this project. Electrofishing reduces growth rates in the long term, though this effect is mitigated in the short term by oxytetracycline injections. There was strong support for initial capture and tagging reducing growth, by an estimated 5.5 mm (see next chapter). The lack of support for an electrofishing effect on growth in this model was not particularly informative, given the confounding of capture method with site. Lack of growth at the Tabragalba site was probably due to a severe algal bloom in the summer months and associated low overnight oxygen levels.

Growth rate varied substantially between locations, a result similar to virtually all analyses of eel growth rates. Eel growth in general is highly variable between individuals, locations, and time periods.

Seasonal variation in eel growth rate in the wild was moderately supported by the model, a result that tallies with the results of the tank experiment, and lends further support to our ageing validation results. Estimated peak growth period in February agrees with the analysis of otolith markings from the ageing validation experiment, which showed least growth during winter. The estimated amplitude of growth variation indicates a 3 to 1 ratio between peak and trough.

These growth rates can be compared with those estimated from otolith analysis, and with those for other eel populations. The population dynamics model estimated average growth of 36.94 ± 2.93 mm per year for Queensland, and 40.77 ± 1.16 mm per year for NSW. The growth rates estimated here are generally in the range of those estimated for other anguillid populations (Aprahamian, 2000), and higher than those generally observed in New Zealand (Jellyman, 1997).

The evidence of both trap avoidance, lower survival following initial capture, and declining population size associated with trapping and electrofishing suggest that mark-recapture population size estimates are likely to overestimate population size. Given this probable bias, population density estimates do not provide a way to transform catch rate indices into population size estimates. Therefore the third objective of this mark-recapture experiment has not been realised.

The low estimated survival rates probably reflect a combination of (in streams) emigration due to the relatively small plots and aversion to the experience of capture, and learning to avoid the gear, and some post-capture mortality; and (in dams) learning to avoid the gear and post-capture mortality. Mortality may have been higher in dams than in streams due to lower levels of oxygen in the water. Some eels not fully recovered from the anaesthetic may have died after sinking to the bottom.

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Chapter 3: Electrofishing reduces growth in longfin eels (*Anguilla reinhardtii*) unless accompanied by injected oxytetracycline

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Abstract

Electroshocking, tagging with passive injected transponders (PIT tags), and injection with oxytetracycline are all used in mark-recapture experiments, but can bias results. In a tank experiment with a factorial design (electroshock x passive injected transponder tags x oxytetracycline), electro-shocked eels grew less than other eels in the short term (three months), except when injected with oxytetracycline. PIT tags and oxytetracycline alone did not affect growth. Beyond three months only the effect of electrofishing on growth rate remained. Injecting eels and possibly other fish captured by electroshocking with oxytetracycline is recommended to reduce capture-related biases in growth estimates over periods of up to three months, but not long term. Electroshocking lowers growth rate in many species, and is known to induce stress, which can lower immune response. Oxytetracycline may reduce rates of infection while the eel immune response is lowered due to electroshocking-induced stress. It may therefore also have benefits from an animal ethics perspective.

Project Objectives

Estimate population parameters required for a management model. These include survival, density, age structure, growth, age and size at maturity and at recruitment to the adult eel fishery. Estimate their variability among individuals in a range of habitats.

Introduction

Field techniques used in ecology sometimes bias research outcomes. Although individual-based experiments such as mark-recapture have many advantages over population-level approaches (Juanes *et al.*, 2000), repeated measurements on the same individuals can introduce bias and so require particular care. We sought to determine whether estimates of growth from a mark-recapture experiment would be affected by the processes of capture (electrofishing) and tagging (Passive Injected Transponder (PIT) tags, and injection with oxytetracycline). We established a tank experiment to examine how these techniques affected the growth of Australian longfin eels (*Anguilla reinhardtii*).

Electroshocking is an efficient method for capturing fish, but is known to reduce growth in a number of species (Dwyer and White, 1995; Snyder, 1995), possibly because of internal haemorrhage, spinal injury (Hollender and Carline, 1994), stress, or tetanising near electrodes which leads to bleeding from gills or vent (Snyder, 1995). Of the species that have been studied, most incur some spinal injury but salmonids are by far the most susceptible (Snyder, 1995). Effects may be seen in both the short and long term (Thompson *et al.*, 1997). Although eel growth rates have previously been estimated from electrofishing mark-recapture studies (Burnet, 1969; Oliveira, 1997), nothing is known of how electrofishing may affect the growth of eels, but their long body profile seems likely to make them susceptible.

Eels appear to be more sensitive than many other fish species to the deleterious effects of tagging (Nielsen 1988). Before PIT tags became available, marking and tagging of eels was generally followed by stunted growth and higher mortality than untreated eels (Berg, 1986; Dekker, 1989; Kuo *et al.*, 1989; Nielsen, 1988). Since Nielsen's review, internal tags have come into widespread

use. Visual Implant Tags have been found not to affect growth, but become unreadable after a year without sacrificing the eel, and have a relatively high 30 per cent loss rate (Bisgaard and Pedersen 1991). PIT tags are now a standard and widely used method for marking individual animals, and have been used in eels (Holmgren and Mosegaard, 1996). Biologically inert and approximately 12 mm by 2.1 mm, they are injected into the animal, do not leave a permanent open wound (Buckley and Blankenship 1990), and are considered to be a low-stress method for marking animals. They have generally been found to have a low impact on growth, e.g. (Quartararo and Bell, 1992), but this has not been validated for eels. The initial injection process has the potential to introduce infection, and the injection process may cause some stress and tissue damage.

Oxytetracycline is used primarily as a marking agent that stains a thin layer of the otolith with colour that shows up as fluorescence under ultraviolet light, and which can be used to validate ageing (e.g. Chisnall and Kalish, 1993). Oxytetracycline may increase the growth of injected fish and bias research results, since it is a broad spectrum antibiotic widely used as a growth promoter in animal husbandry (Gaskins *et al.*, 2002). However, the process of injection may retard growth by causing stress, damaging the injection site by the high local concentration of OTC, or introducing viral infections at the puncture wound, especially if needles are reused. OTC has been demonstrated to have toxic effects in some species (Marking *et al.*, 1988; Toften and Jobling, 1996). Validation in individual species is recommended (Natanson *et al.*, 1999).

Interactions between these three treatments may also occur, because the stress of receiving a number of treatments may retard growth more than the sum of individual treatment effects. On the other hand, the antibiotic effect of injected oxytetracycline may reduce the impact of other treatments, such as injecting PIT tags or electroshocking. Growth promotion by OTC relative to tagged sharks without OTC has been reported in sharks (Simpfendorfer, 2000). Accordingly, this experiment was designed to see whether such interactions arise and to explore the biases that might be introduced as a consequence. We show that oxytetracycline can be beneficial in covering for the short term adverse effects of electroshocking.

Methods

The basis of the methods described below is that eels were allocated to eight treatment groups (electroshock x PIT tag x oxytetracycline), all of which also were individually identified with visual implant elastomer (VIE) tags, and one control group without VIE tags or any treatments. Different size classes were placed in separate tanks.

VIE tags are a fluorescent elastomer material that is injected as a liquid and cures within hours into a pliable biocompatible solid. The material is injected beneath transparent or translucent tissue. Due to its flexible liquid nature, the small size of the tag, and the lack of a continually open wound such as that left by an anchor tag, it is a very low stress method for marking fish with little effect on their subsequent survival. It has been used on fish as small as 8 mm standard length with low mortality (Frederick, 1997).

Eels used in the experiment were obtained from streams in southeast Queensland during 1999 by electrofishing and trapping. In general, the smaller eels were captured by electrofishing and the larger eels captured in traps.

After several months in holding tanks, 306 eels were anaesthetised for treatment by immersion in one part clove oil to approximately 40 000 parts freshwater. They were then divided into four similar size-class groups in order to reduce harassment and cannibalism: 18–27 cm, 27–41 cm, 41–62 cm, and 62–93 cm (q.v. Knights, 1987). Within each size class the eels were divided into two groups, with four ninths in one group (electroshock) and five ninths in the other (no electroshock).

Several days after recovery from the anaesthetic the electroshock group were subjected to five seconds of 200 volt DC electric current at 120 Hz, from a Smith-Root backpack electrofisher.

Eels were then re-anaesthetised. Electroshocked eels were randomly assigned within size class to one of four treatment groups (PIT tag or no PIT tag x oxytetracycline or no oxytetracycline), and all were uniquely identified with VIE tags. Non-electroshocked eels were randomly assigned within size class to the above four treatment groups (PIT tag or no pit tag x oxytetracycline or no oxytetracycline), all of which were uniquely identified with VIE tags. All eels were weighed and measured.

PIT tags were injected into the back of each eel just to the left of the start of the dorsal fin. Tags were injected to lie in the layer of muscle close to the skin surface. OTC was injected intramuscularly into the tail of the eel at approximately 2/3 of the length of the eel, at a dosage of 75 milligrams per kilo of body weight.

Eels were held in four 4000 litre tanks, with flow-through bacterial filter; shade-cloth and water supply monitoring system (ammonia) + removal of food waste by siphoning from tank bottoms 24 hours after feeding. Eels were supplied with hiding areas/habitat enrichment using plastic pipes, at diameters suitable for the size range in each tank.

Eels were maintained in the tanks for one year. Temperature was allowed to vary with ambient temperature throughout the year. Mortalities were counted and removed daily. All eels were fed approximately three times per week, initially with fish and later with eel food pellets. Frequency of feeding was reduced to twice or once per week during winter when demand was lower.

After three months and subsequently every two months, all eels were anaesthetised using clove oil, weighed and measured, and tags were examined. VIE tags were verified using an ultraviolet light and amber glasses.

In analysing the data we compared growth increments between the beginning of the experiment and weighing 2, between weighings 2 and 3, and between weighings 3 and 4.

Growth increments were transformed in order to remove heteroscedasticity, using the transformation $\text{trans_length} = (\text{length} + \text{constant})^{0.5}$. The constant was chosen to be half the size of the discrete unit minus the most negative value (Yamamura, 1999). Relative size within tank at the start of the period was calculated as the difference between the size of the eel and the average size within the tank.

Data were modelled using a generalised linear model with normal errors in Genstat 5 Release 4.1 with the factors tank, pit tag, electroshock, OTC, and the covariate relative size. All interactions up to 5 way were investigated. Data were also modelled at the individual tank level.

Results

Growth during period 1

The transformation $y = \sqrt{x+12.5}$ was applied to eel growth. All eels from tanks A, B, C and D were included in the analysis.

Two interactions were statistically significant: the interaction between OTC and electrofishing, and between relative size and tank (Table 1). The average growth for individuals that received OTC was the same, whether or not they were electrofished, and this growth rate was similar to that of individuals that received neither of these treatments. However, individuals that received electrofishing but not OTC grew significantly less on average (Table 2).

In tanks A and B (size classes 18–27 and 27–41) growth was positively correlated with initial size, but in tanks C and D (size classes 41 to 62 and 62 to 93) growth was negatively correlated with initial size (Table 3).

Table 1: Accumulated analysis of variance table for growth during period 1, based on type 3 sums of squares. Growth has been transformed by $y = \sqrt{x+12.5}$.

Change	d.f.	s.s.	m.s.	v.r.	F pr.
+ relsz[0]	1	1.085	1.085	7.98	.001
+ OTC	1	3.733	3.733	3.36	0.068
+ Tank	3	58.714	19.571	17.62	<.001
+ Electrofished	1	10.596	10.596	9.54	0.002
+ relsz[0].OTC,	1	1.156	1.156	1.04	0.309
+ relsz[0].Tank	3	27.542	9.181	8.26	<.001
+ OTC.Tank	3	1.113	0.371	0.33	0.801
+ relsz[0].Electrofished	1	1.164	1.164	1.05	0.307
+ OTC.Electrofished	1	7.538	7.538	6.79	0.010
+ Tank.Electrofished	3	7.850	2.617	2.36	0.072
Residual	246	273.261	1.111		
Total	264	423.752	1.605		

Table 2: Estimated mean values for growth showing the interaction between electrofishing and OTC during the first growth period. The prediction is based on a fixed value of 20.65 for the relative size. The predictions were standardised by averaging over the values of all tanks. The standard errors are appropriate for interpreting the predictions as summaries of the data rather than as forecasts of new observations.

	Not electrofished		Electrofished	
	Prediction	S.e.	Prediction	S.e.
No OTC	4.883	0.136	4.144	0.135
OTC	4.761	0.137	4.716	0.128

Table 3: Estimated mean values of growth showing the interaction between release size and tank during the first growth period. The predictions have been standardised by averaging over the levels of electrofishing and OTC. The standard errors are appropriate for interpreting the predictions as summaries of the data rather than as forecasts of new observations.

Tank	Relative size	Prediction	Std. Error
A	0	4.570	0.164
	50	5.303	0.221
B	0	4.383	0.130
	50	4.748	0.136
C	0	5.084	0.119
	50	4.795	0.163
D	0	3.638	0.268
	50	3.372	0.199

Growth during period 2

The transformation $y = \sqrt{x+7.5}$ was applied to eel growth. Eels from tank A and those without PIT tags were omitted from the analysis, because loss of VIE tags meant eels without PIT tags were unidentifiable, and most of the smaller eels in tank A had lost their PIT tags.

The interactions between OTC and electrofishing, and the three way interaction between OTC, electrofishing, and Tank were not statistically significant (Table 4).

There was a statistically significant interaction term between electrofishing and tank (Table 4), with a larger negative effect on growth by electrofishing in Tank B (Table 5). Eels in the other tanks also grew less when electrofished, but the effect was much smaller. Eels in Tank B grew the most, followed by Tank C, then Tank D.

Table 4: Accumulated analysis of variance table for growth during period 2, based on type 3 sums of squares. Growth has been transformed by $y = \sqrt{x+7.5}$.

	d.f.	s.s.	m.s.	v.r.	F pr.
+OTC	1	0.977	0.977	0.41	0.522
+Tank	2	42.501	21.250	9.01	<.001
+Electrofished	1	17.401	17.401	7.38	0.008
+resize2	1	3.579	3.579	1.52	0.221
+Tank.Electrofished	2	23.361	11.680	4.95	0.009
+resize2.Tank	2	9.777	4.888	2.07	0.132
Residual	92	216.982	2.359		
Total	101	314.577	3.115		

Table 5: Estimated mean values of growth showing the effect of electrofishing and variation between tanks during the second growth period. The prediction is based on a fixed value of 20.65 for the relative size. The standard errors are appropriate for interpreting the predictions as summaries of the data rather than as forecasts of new observations.

		Tank B		Tank C		Tank D	
OTC	E'fished	Prediction	S.e.	Prediction	S.e.	Prediction	S.e.
0.00	0.00	6.654	0.396	4.361	0.384	4.019	0.566
	1.00	4.638	0.330	4.097	0.390	3.835	0.608
1.00	0.00	6.930	0.398	4.638	0.388	4.296	0.576
	1.00	4.915	0.329	4.374	0.405	4.112	0.595

Growth during period 3

The transformation $y = \sqrt{x+12.5}$ was applied to eel growth. As for the previous analysis, eels from tank A or without PIT tags were omitted from the analysis.

Statistically significant effects were Tank and electrofishing (Table 6). Tank B eels grew considerably more than those in other tanks, and electrofished eels grew less (Table 7).

The data were re-analysed with six outliers omitted. These individuals grew little, if at all, during the entire experiment, and five of them had been electrofished. With the slow-growing outliers removed, there was still a significant electrofishing effect, retarding growth.

Table 6: Accumulated analysis of variance table for growth during period 3, based on type 3 sums of squares. Growth has been transformed by $y = \sqrt{x+12.5}$.

Change	d.f.	s.s.	m.s.	v.r.	Fpr.
+relsize2	1	0.061	0.061	0.03	0.874
+OTC	1	4.818	4.818	1.97	0.164
+Tank	2	137.913	68.957	28.23	<.001
+E'fished	1	15.246	15.246	6.24	0.014
Residual	90	219.829	2.443		
Total	95	377.868	3.978		

Table 7: Estimated mean values of growth showing the effect of electrofishing during the third growth period, with variation between tanks. The prediction is based on a fixed value of 20.65 for the relative size, and have been standardised by averaging over the levels of OTC. The standard errors are appropriate for interpreting the predictions as summaries of the data rather than as forecasts of new observations.

Electrofished	0.00		1.00	
	Prediction	S.e.	Prediction	S.e.
Tank				
B	8.249	0.335	7.403	0.272
C	5.809	0.293	4.963	0.301
D	5.285	0.432	4.440	0.445

Discussion

Electrofishing and OTC

Eels that were electrofished tended to grow less than other eels, although OTC injection alleviated this effect during the first three months. That eel growth is affected by electrofishing has not previously been demonstrated, although it has been observed in a number of other species. Electroshock causes stress and raises cortisol levels (Barton and Grosh, 1996), which lowers immune responses (Pankhurst and van der Kraak, 1997, VanderKooi *et al.*, 2001). Physiological stress may also reduce growth by lowering thyroid hormone levels (Pankhurst and van der Kraak, 1997; Redding *et al.*, 1986). Antibiotics may improve the efficiency of animal growth by inhibiting the normal microbiota, increasing nutrient use and reducing the maintenance costs of the gastrointestinal system (Gaskins *et al.*, 2002). Part of electrofishing's effect on growth may stem from reduced ability to control gut flora or other infective agents, as a result of shock-induced stress. Oxytetracycline may defend the organism against infection during this period.

The ability of OTC to reduce the effects of electrofishing on growth, in the short term, has not previously been described for any species, but higher growth rates have previously been found in tagged dusky sharks with OTC than in those without OTC (Simpfendorfer, 2000). If its action is to alleviate the immediate effects of stress, it may have similar benefits if injected into fish in tag-release programs using other methods, such as line fishing.

Electrofishing appeared to reduce growth rates throughout the experiment, perhaps due to the same type of skeletal damage that has been observed in salmonids. Thompson *et al.*, (1997) found reduced growth and condition of rainbow and brown trout one year after electroshocking. We did not examine the eels internally for damage, but no external features such as obvious bends in the spine were observed. Certainly the long body shape of eels would potentially make them more susceptible to effects of electrofishing than most other Australian freshwater fish species. The effect does not appear to be exclusive to a few badly damaged eels, since when the six low-side outliers were removed from the six to eight month period, a significant reduction of growth rate with shocking was still apparent.

Although electroshocking had long-term effects, OTC did not appear to provide long-term protection. If OTC's action was to alleviate the immediate effects of stress and lowered immune response, why electrofishing related stress re-emerge after the three month protection period provided by OTC is not clear, unless some internal physical damage was affecting immunity. However it is possible that lack of power in the long-term analysis due to lower sample sizes meant that a real interaction was not statistically significant.

For animals that were not electroshocked, OTC did not have any apparent effect. This may be because the eel immune system was not compromised by the other treatments, so OTC could have no compensatory effect. All eels would have been stressed to some extent by the process of capture and anaesthetisation, but we speculate that these stresses were not sufficient to lower their immune response.

Some of the eels had initially been captured using electrofishing, but we did not expect this to affect our results. Eels were randomly distributed among treatments with respect to initial capture method, and repeated electrofishing is likely to produce a cumulative effect. (Gatz *et al.*, 1986) found that repeated electrofishing reduced the instantaneous growth rates of brown and rainbow trout, and that this effect increased with the number of times the fish was electrofished. Eels electrofished in the confines of the tank for this experiment tank may have received a higher concentration of electrofishing power, than eels electrofished in open wild conditions.

PIT tags

PIT tags appear to be an effective low-impact method for tagging eels, unlike almost all other methods. There was no difference in growth between PIT tagged and untagged eels, independent of other treatments. Tissue damage, stress, and infection associated with tag injection therefore do not appear to be significant problems, given that needles were sterile and used only once. Since PIT tags are internal, and the entry wound is able to heal, it lacks the weakness of other tags that leave an open wound for entry of infection. Due to their similarities, this conclusion may be extended to coded wire tags (q.v. Thomassen, *et al.*, 2000).

PIT tags provide functionality not found in coded wire tags and Visual Implant Tags, which have also been found not to retard growth (Bisgaard and Pedersen, 1991; Thomassen *et al.* 2000), since they allow long-term identification of the eel without sacrificing recaptured animals. In this experiment a significant proportion of visual implants had become invisible or were lost after six months. Coded wire tags are usually only read after sacrificing the animal, although they can be detected with an electronic wand, which gives some scope for batch tagging using different body locations.

PIT tags, on the other hand, do not appear to interfere with growth, unlike almost all other methods for individually identifying eels.

Relative size

In tanks without refuges, aggression between eels can force small eels to swim constantly, and also reduce their access to food. Thus they can grow more slowly than larger eels (Wickins, 1987, Yahyaoui, 1988). We attempted to reduce such competition by providing plastic pipes of various sizes, which although not eels' habitat of choice (Glova, 1999), enables smaller eels to take refuge.

For the first three months in the two tanks with small eels, larger eels grew more than smaller eels. The opposite pattern was observed in the two larger size-class tanks. This difference may have occurred because space competition was more significant among the smaller eels, or because spinal injury occurred in the larger eels within each of the large tanks. Spinal injury can be positively correlated with fish length (Dalbey *et al.*, 1996). Later in the experiment no significant effects were associated with relative size, but power was lower due to halving of sample size.

Tank/eel size

Different growth rates between the tanks were inevitable since they contained different sizes of eel, which were fed different types and sizes of food. Also the arrangement of the tanks and filters meant that water quality and temperature varied between the tanks. Different-sized eels were electrofished separately, and despite care may have experienced different levels of shock.

What are the practical implications?

This result adds to the body of evidence that tag-release experiments using electrofishing are not ideal for estimating growth rates. Injection with OTC on capture appears to initially reduce the stunting of growth, but not in the long term. Other capture methods such as fyke netting (Rossi *et al.*, 1988) or cage trapping (Bozeman *et al.*, 1985) may allow tag-release experiments to give better estimates of growth rate. The stress of trapping, handling, and anaesthetisation may also stunt growth compared to untreated animals, but our experiment did not address this issue since all animals were anaesthetised and handled.

Population-level methods such as otolith reading and back-calculation (e.g. Panfili and Ximenes 1994), rather than methods based on individual recapture, avoid biases due to repeated capture. However, these methods introduce other potential sources of bias such as inaccurate otolith grinding

(Panfili and Ximenes, 1992), ageing error, or spatial, temporal, and individual variation in the ways annuli are laid down.

Some studies have shown that otolith reading and mark-recapture based on non-electrofishing methods give similar estimates of growth rate (Poole and Reynolds, 1996).

Conclusion

Pit tagging appears to be an effective way of tagging eels without affecting their growth. However, capturing eels using electrofishing is likely to affect their growth in the long term. This growth retardation may be alleviated in the short term by injection with oxytetracycline.

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Chapter 4: Ageing validation for longfinned eels *Anguilla reinhardtii* from sub-tropical Queensland

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Abstract

Oxytetracycline was used to mark the otoliths of captive and wild eels for validation of periodicity of otolith check mark formation. Complete check marks were found to form annually in over 80 per cent of longfinned eels in sub-tropical Queensland. Timing of annulus formation was variable but the majority of annuli were formed in the winter spring period, with a small proportion being formed in summer. The proportions of wild eels with greater or lesser periodicity of otolith formation were equal. It therefore appears to be acceptable to use check marks from *A. reinhardtii* otoliths for age based models.

The relationship between somatic growth and otolith growth is strongly linear in wild eels, therefore otoliths can be used for back calculation of length at age. However there is evidence for uncoupling of otolith and somatic growth relationships in aquacultured eels or in populations with rapid growth rates.

Project objective

Estimate population parameters required for a management model. These include survival, density, age structure, growth, age and size at maturity and at recruitment to the adult eel fishery. Estimate their variability among individuals in a range of habitats.

Introduction

Australian longfinned eels (*Anguilla reinhardtii*) are important predators in eastern Australian freshwater ecosystems. They also support small scale fisheries throughout their range. These include a commercial trap fishery in weirs, large impoundments and farm dams in Queensland; a larger scale trap fishery in tidal waters of New South Wales (NSW) (Pease *et al.*, 2003); and fisheries in Victoria and Tasmania. In Queensland glass eels are harvested for grow out in the aquaculture industry.

Ageing validation is necessary for credible assessments of longfinned eel population dynamics. Estimates of age at maturity, growth and mortality rates are required for assessing stocks of longfinned eels. These are likely to vary between the different climatic regions within their distribution. Determining the age of eels is key to estimating growth, age at maturity and mortality. Validation of ageing methods is therefore critical. Otoliths markings have been widely used around the world to age *Anguilla* species other than *A. reinhardtii* (Chisnall and Kalish, 1993; Graynoth, 1999; Jellyman, 1979; Panfili *et al.*, 1994; Oliveira, 1996; Pool and Reynolds, 1995; Svedang *et al.*, 1998; Vollestad, 1985). Sloane (1984) inferred annual periodicity of checks in otoliths from *A. reinhardtii* collected from a cool temperate Tasmanian catchment. Sloane used marginal increment analysis, but this methodology can be unreliable (Campana, 2001). Recently Shiao *et al.* (2002) validated formation of a freshwater check mark in otoliths of glass eels of *A. reinhardtii* from south-eastern Queensland and NSW. This mark occurs near the marine nucleus, an opaque mass in the centre of the otolith. Pease *et al.* (2003) validated formation of annual checks, including the first annulus in oxytetracycline marked otoliths of *A. reinhardtii* from NSW. However, Pease *et al.* (2003) also noted the formation of incomplete or false checks. False checks have been reported in other eel species too, including *A. rostrata* (Oliviera, 1996), *A. anguilla*

(Tzeng *et al.*, 1994), *A. australis* and *A. dieffenbachia* (Graynoth, 1999). False checks observed by Pease *et al.* (2003) were narrower and less distinct than annuli and did not extend continuously around the otolith.

No previous studies have validated *Anguilla* annuli from sub-tropical regions, where growth increments are likely to be less distinct and may form during summer. Daily growth increments have been validated for two species of tropical eel, *A. celebesensis* (Arai *et al.*, 2000) and *A. marmorata* (Sugeha *et al.*, 2001), but annulus formation is a different matter. Growth rates of eels in subtropical regions are likely to slow less during winter than for eels from more temperate zones. There is also anecdotal evidence from eel fishers that feeding activity of longfinned eels slows during the hottest periods of the south-east Queensland summer, when water temperatures may reach more than 30°C.

If proportionality between otolith size and eel total length can be demonstrated, then validation of annual increments will also enable back calculation of growth rates. This allows for estimation in variability in growth and recruitment to the fishery within and between different habitats. Validating the age and back-calculating growth rates of Queensland populations of eels will be a key step in meeting the project objective (see above) of estimating population parameters required for a management model and estimating their variability among individuals in a range of habitats.

The aims of this study are to validate that annual increments form in the otoliths of south-east Queensland long finned eels and to establish that there is proportionality between fish length (growth) and otolith incremental distances.

Methods

Tank held eels

An experiment was conducted on tank held eels to examine otolith increment periodicity of four size classes of eels. This experiment formed part of a larger experiment that examined the effect of passive integrated transponder (PIT) tags, electrofishing, and oxytetracycline (OTC) injection on growth. This data is reported elsewhere in this report. The OTC marked eels were also used to validate increment periodicity. We caught 306 eels from streams in southeast Queensland during 1999. Eels were stocked into four 4000 litre tanks in similar size-class groups in order to reduce conflict and cannibalism. They were graded into size classes A, B, C and D: 18–27 cm, 27–41 cm, 41–62 cm and 62–93 cm, based on Knights (1987). There were 65, 104, 99 and 40 eels in each respective size class.

In July 1999 eels were selected at random from each group for injection of OTC. A total of 29 eels from size class A, 46 eels from size class B, 44 eels from size class C and 18 eels from size class D received OTC injections. OTC leaves a mark in bony tissue of fish. These marks fluoresce under UV light and can be used as reference marks to validate formation of annual increments in otoliths. Eels to be marked with OTC were first anaesthetised in one part clove oil to 40 000 parts water. OTC was then injected into the musculature of selected eels, approximately halfway between the vent and the tail tip, at a dosage of 75 milligrams per kilo of body weight. Each OTC injected eel was uniquely marked with visible implant fluoro-elastomer (VIE) tags for later identification. VIE tags are injected as a liquid and soon cure into a pliable biocompatible solid. The material is injected beneath transparent or translucent tissue. Due to its flexible liquid nature, the small size of the tag, and the lack of a continually open wound such as that left by an anchor tag, it is a very low stress method for marking fish with little effect on their subsequent survival. It has been used on fish as small as 8 mm standard length (Frederick, 1997). A randomly selected sub-group of the OTC marked eels was also tagged with PIT tags.

PIT tags are small 1.5 mm x 5 mm magnetically coded tags, which can be injected into the musculature or under the skin. Fourteen eels from group A, 22 from group B, 21 from group C and 8 from group D received PIT tags. PIT tags were injected into the back of each eel just to the left of the start of the dorsal fin, with the hypodermic needle supplied with the PIT tags (Avid PIT tags).

Eels were held in four 4000 litre tanks, with a recirculating bio-filter system, shade-cloth cover and ammonia monitoring system. All eels were fed approximately three times per week, initially with fish and later with eel food pellets. Frequency of feeding was reduced to twice or once per week during winter when demand was lower. Food waste was removed by siphoning from tank bottoms 24 hours after feeding. Fresh UV treated dechlorinated water was used to top up tanks after siphoning.

Eels were supplied with hiding areas constructed from plastic pipes, at diameters suitable for the size range in each tank. Eels were maintained in the tanks until November 2000. Temperature was allowed to vary with ambient temperature throughout the year. Temperatures ranged from approximately 12°C to 30°C, although malfunction of a tank temperature data logger has prevented us from knowing the precise range of temperatures experienced in the tanks. Tanks were checked for eel mortalities daily. In November 2000 all remaining eels in the tanks were captured and euthanased in an overdose of clove oil. Euthanased eels were identified, measured and then frozen for later processing of otoliths (see below).

Wild eels

Sites were established for mark-recapture experiments for eels (see Figure 1 Chapter 2) between Beaudesert and Elimbah, in the greater Brisbane area of south-east Queensland. Selected sites were lentic (Jess Dam, Tabragalba Lagoon, and Sherwood Arboretum Pond) and lotic (lower Kedron Brook at the Grange, Upper Kedron Brook at Keperra and Samford Brook). Sites ranged from approximately 12 km to 36 km inland from the nearest point on the coast. Middle thread distances upstream from tidal influences were at least twice the straight-line distance to the coast.

The field ageing validation experiment was designed to double up as a mark-recapture experiment for estimating the abundance/density of eels (see Chapter 2) and for field based ageing verification. The experiment carried out over a period of two and a half years, from July 1999 to February 2001. None of the sites were fished commercially during the experiment. At the start of the experiment a temperature logging device was attached to a tree root or other fixed point in a position where it would remain underwater for the duration of the experiment.

Each sampling event consisted of a two-day session at each site. At stream sites a stopper net was set at each end of sites where closure was not already complete. Fyke nets and baited traps were set on the first evening at the site. The following morning the traps were cleared. All captured eels were immediately anaesthetised with clove oil, measured, uniquely tagged and injected with oxytetracycline (OTC) at a rate of 75 mg/kg body weight. OTC was used to place a mark in otoliths (see tank eels above). Eels over 25 cm were tagged with internal PIT tags. Eels smaller than 25 cm were injected with visual implant elastomer in three of eight positions, in a unique combination of three colours. After recovery from anaesthesia, eels were released within 20 metres of the capture site. The site was then electrofished using the standard 10 metre shots. Eels captured by electrofishing were again measured, marked and released as above. All recaptures were recorded but not remarked.

After completion of electrofishing fyke nets were again set overnight, followed by a second electrofishing session. Stopper nets were removed after this electrofishing session. This gave

a four-stage closed mark-recapture experiment in two days of sampling per site. Eels caught more than once over the two day period were not given a second injection of OTC. Lagoon and small dam sites were also sampled over two days and treated as closed sites. 20 traps baited with pilchards were set throughout lentic sites at randomly spaced distances on the first night. These were pulled the following morning, then reset, and spaced at a new set of random distances. Captured eels were measured and marked and released as above.

The entire sampling process was repeated at intervals of six months. On these occasions, recaptured eels were given another injection of OTC to assist with the ageing verification process. The final OTC marking of eels was completed in January 2001. In February 2001 all sites were sampled for a final time. Recaptured tagged eels on this occasion were retained, euthanased by an overdose of clove oil and taken back to the laboratory for removal and processing of otoliths. Any recaptured eels which died in traps (from low overnight oxygen levels) during the course of the Mark-Recapture experiments, were retained, and taken back to the laboratory for processing of otoliths.

Otolith preparation

All eel specimens were blast frozen and stored at -24°C until thawed for processing. The sample number, tag ID, tank or catch location, total length and sex were recorded for each eel. Sagittal otolith pairs were then removed from each specimen. Otolith pairs were rinsed clean, dried and then stored in labelled airtight plastic vials.

The best condition otolith of each pair was chosen for sectioning. If both otoliths were in comparable condition a left or right otolith was selected at random for sectioning. Each selected otolith was imbedded in a polyester resin block formed using latex moulds. A Buehler Isomet[®] low speed saw fitted with a diamond wafering blade was used to cut transverse sections through the core of the otolith.

Sections ranging in thickness from 200 to 450 μm were compared initially to determine the optimum thickness with regard to ease of handling and readability. Sections of 300 μm were found to be the most appropriate. Immediately after cutting each section was examined under a stereo microscope. If it was of unsatisfactory quality, cutting was continued until the best possible section was obtained. After rinsing and drying, the section was mounted on a labelled glass slide under a cover slip, using polyester resin as the mounting medium.

Otolith reading

To determine the periodicity of otolith formation, otoliths were viewed at 40x magnification using a combination of reflected and UV light under Leica MZ6[™] or WildMZ3[™] binocular and Nikon MicrophotFXA[™] UV microscopes, with a Pulnix TMC6[™] colour digital camera attachment. The camera was linked to a computer screen where the image was transmitted for viewing. When viewed under natural light, otoliths were examined for annuli. When viewed under UV light, OTC marks in the otoliths fluoresced and became visible. Both UV and reflected light images of otoliths were captured with a screen grabber in the image analysis program Optimas 6.1[™] and saved as jpeg files for later viewing and analysis (Figures 1 and 2).

For samples viewed under reflected light, a readability index was applied. These were as follows

1. unreadable
2. interpretable, but not confident
3. multiple interpretations possible
4. readable, but not totally confident
5. readable, totally confident.

Age was estimated by counting annuli (opaque zones) between the nucleus and outer margin. The freshwater check was separated from annuli based on the descriptions by Shiao *et al.* (2002) and was assumed to be the first mark after the marine nucleus. Each otolith was read independently on two separate occasions. Opaque zones were read as an annulus if they extended continuously around the otolith. Other marks which were less distinct and which did not extend continuously around the otolith were considered to be false annuli or supernumerary checks. These were observed in some specimens but not recorded as annuli. For reflected light images distance was measured in mm (to three decimal places) along the ventral axis from the centre of the nucleus to each annulus and to the edge of the otolith (Figure 3). This was done using Optimas 6.1™ image analysis software, adjusted for the 40 times magnification of the image. Similarly the distance from the centre of the nucleus to any fluoresced OTC mark was measured on the UV light images.

Validation of annuli

The position of OTC marks was recorded relative to the position of the *nearest check mark* assumed to be an annulus. The distance between annuli before and after an OTC mark was divided into four equal zones, numbered 1 to 4. Zones occurring before an annulus were designated 1A, 2A, 3A and 4A, zones occurring after an annulus were designated A1, A2 etc. Zones between an annulus and the edge of the otolith were designated A1E, A2E, A3E and A4E. The translucent zone of the annulus was designated A. Therefore, OTC mark positions in relation to the nearest ‘supposed annulus’ were always recorded in either zone 3A or 4A (before formation of an annulus), A (on the annulus), A1, A2 (after the annulus), or in A1E through to A4E (between the annulus and the edge of the otolith). As the dates of OTC injection and sacrifice were known, the relative position of the OTC mark was used to estimate the approximate time of formation of the nearest check mark. In some wild mark-recaptured eels there was more than one OTC mark. Each OTC mark position was analysed separately to see if results were consistent. The number of expected annuli to form after this mark was estimated based on assumption of either winter-spring annulus formation or summer-early autumn annulus formation. Comparison of numbers of observed and expected annuli post the OTC mark or marks was used to verify whether or not complete checks were actually annual checks.

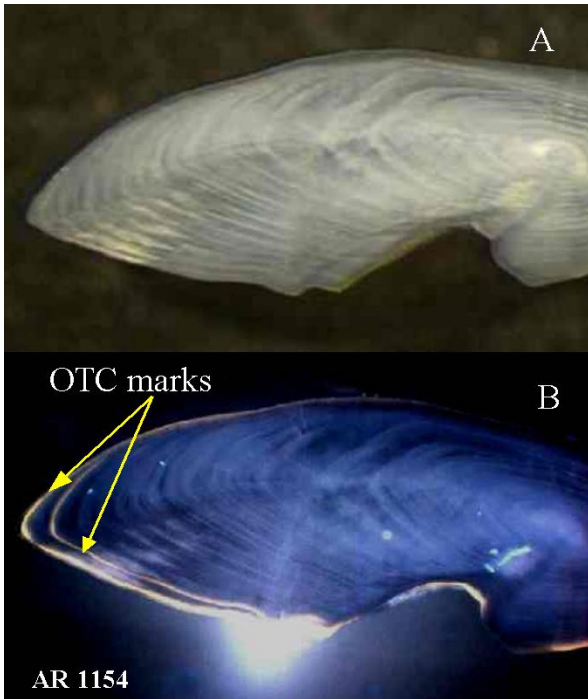


Figure 1: An otolith from a twice OTC marked and then recaptured wild eel viewed under reflected light (A) and UV light (B) with two OTC marks visible in the UV image.

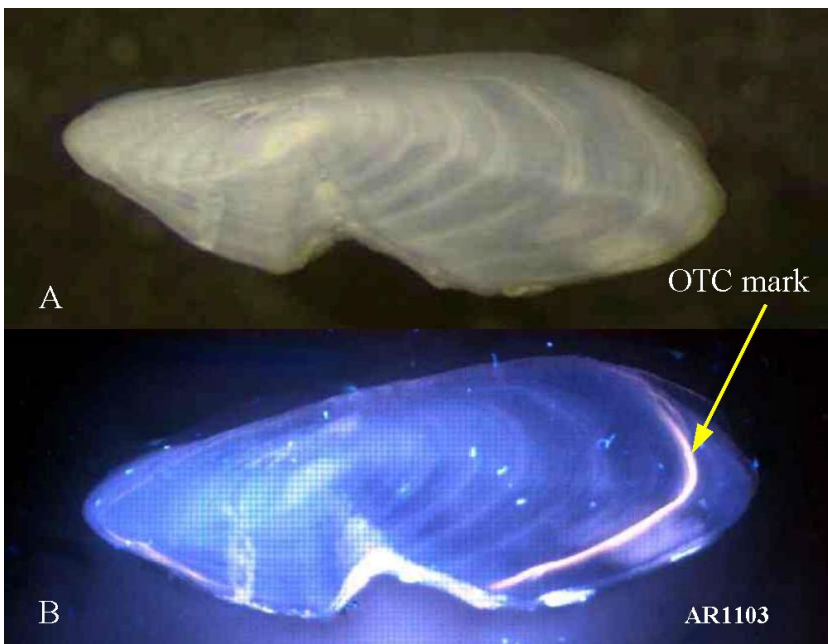


Figure 2: An otolith from a tank held OTC marked eel viewed under reflected light (A) and UV light (B) with a single band of OTC visible in the UV image.

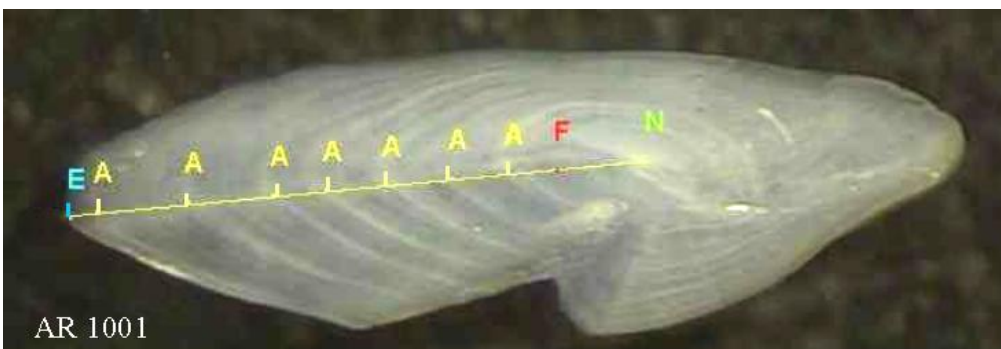


Figure 3: An otolith from *Anguilla reinhardtii* viewed under reflected light, showing annuli (A) the marine nucleus (N), freshwater check (F) and the edge of the otolith (E). Distances were measured along the axis as marked from the centre of the marine nucleus to the edge.

Relationship between otolith length and total length of eels

A comparison was made between total length of eels and the length of otoliths measured from the centre of the nucleus to the edge of the otolith along the ventral axis. A strong linear relationship would mean that reliable estimates of length at age could be made through back calculation.

Otoliths were measured along the ventral axis in mm to three decimal places as outlined in ‘*Otolith reading*’ above. Total lengths of eels were measured in mm at the time of otolith extraction. However, field measurements of eels made between one day and three months before the death or euthanasing of these eels were often greater than those of the same animal at time of otolith extraction. It therefore appeared that some shrinkage was occurring post-death of the eel. The amount of shrinkage did not appear to be related to the size of the eel. Mean shrinkage was 14.7 mm. Morison *et al* (2003) found post-mortem shrinkage to be a problem in four species of temperate and tropical marine fishes they examined, and for three of the four species, shrinkage was not related to the size of the fish. We used the mean shrinkage of 14.7 mm as a correction factor for processed eels. Nevertheless as an alternative to correction for shrinkage, otoliths were also measured to the OTC mark in mm. This measure was compared with the total length of eels as measured in the field at the time of OTC injection.

A simple linear regression was then run in Genstat™. Otolith length was set as the independent variable and total length as the dependent variable. Regressions were run using both the corrected measures for shrinkage, and the field measures at time of OTC injection. Data from tank and wild eels were analysed separately and as pooled data in this analysis. Slopes and intercepts of the regressions were compared for significant differences using the simple linear regression with groups option in Genstat™.

Results

Timing and periodicity of check formation

Of the 137 OTC marked eels held in tanks, only 81 were used for validation of ageing. Some otoliths were unreadable (score of 1), others had no detectable OTC mark under UV light and some eels were lost, either through escape from tanks or cannibalism of dead eels before they could be cleared from the tank. There was also loss of tags, making identification of some individual eels difficult. The latter two problems were mostly in tank A, where PIT tag retention was poor in the smallest size class of eels, and where size differentiation increased with time, whereby larger more dominant individuals were able to cannibalise slower growing subordinate eels. Loss of readability of VIE tags was also a problem across all tanks (see Chapter 3).

A total of 560 eel captures were made throughout the mark recapture process, comprising, 437 individual eels. However only 35 eels with visible OTC marks and readable otoliths (score >1) comprised the combined February catch of recaptured eels and recaptured eels which had died in the course of the mark recapture work. These 35 eels were used for the field validation of ageing.

Table 1: Position of OTC mark relative to nearest check in 81 tank held eels and estimated time of check formation.

	Position of OTC mark relative to nearest check				
	3A	4A	A	A1	A2
Estimated time of check formation	September-November	August-September	July	May-June	March-May
Count	9	22	36	12	2
Per cent total	11.1	27.2	44.4	14.8	2.5

Table 2: Number of subsequent check marks following check nearest to OTC mark in 81 tanks held eels. If checks are formed annually (regardless of whether checks are formed in summer or winter-spring), expected number of additional marks is 1. Eels were tagged July 1999 and sampled November 2000.

	Number of subsequent check marks		
	0	1	2
Count	10	69	2
Per cent total	12.3	85.2	2.5
Expected per cent	0	100	0

Tables 1 and 2 show the ageing validation results for the tank held eels. Table 1 shows estimated time of check formation. Most checks are estimated to have formed between May and September with 44 per cent corresponding closely to the OTC mark, which was injected in July. Table 2 compares the number of subsequent checks formed in the period following the formation of the check closest to the OTC mark, as compared to the expected number of checks if formation was annual. Approximately 85 per cent of otoliths from tank held eels had check formation corresponding with an annual one, with just over 12 per cent have less than annual formation.

Table 3 shows position of check marks relative to the OTC marks in wild eels, with estimated timing of check formation and number of subsequent checks to form in the period the eels were at large, compared with expected number to form should checks be formed in summer autumn or winter spring. Table 4 summarises these data and shows that almost 83 per cent of eels had checks form in winter-spring, with the balance having checks that formed in summer-autumn.

Approximately 83 per cent of eels were found to have checks forming on an annual basis, with the balance having check formation rates evenly split between less than annual and greater than annual.

Table 3: Position of OTC mark relative to nearest check in marked-recaptured wild eels with estimated time of check formation and counts of observed subsequent checks formed compared with number of subsequent checks expected to form.

Site	Date tagged	Final recapture date	Time at large months	Position of UV mark	Estimated time of formation of check nearest to OTC mark	Additional checks post first OTC mark	Expected additional checks if winter-spring/summer-autumn
Sherwood	18/1/2001	Feb 2001	1	A4E	Sept-Nov 2000	0	0/1
Sherwood	5/5/2000 2/11/2000	Feb 2001	9.5 3	4A A (stress?)	June-July 2000 Nov 2000	1	0/1
Jess Dam	19/1/2001	Feb 2001	1	A4E	Sept-Nov 2000	0	0/1
Jess Dam	19/1/2001	Feb 2001	1	A4E	Sept-Nov 2000	0	0/1
Jess Dam	19/1/2001	Feb 2001	1	A4E	Nov-Dec 2000	0	0/0
Jess Dam	19/1/2001	Feb 2001	1	A4E	Aug-Oct 2000	0	0/1
Jess Dam	19/1/2001	Feb 2001	1	A4E	Sept-Nov 2000	0	0/1
Jess Dam	15/10/2000 19/1/2001	Feb 2001	4 1	A A4E	Oct 2000 Oct 2000	0	0/1
Jess Dam	19/1/2001	Feb 2001	1	A4E	Sept-Nov 2000	0	0/1
Jess Dam	19/1/2001	Feb 2001	1	A4E	Sept-Nov 2000	0	0/1
Jess Dam	??/4/2000	Feb 2001	10	4A	May-Jul 2000	0	0/1
Jess Dam	19/1/2001	Feb 2001	1	A4E	Aug-Oct 2000	0	0/1
Keperra	14/7/2000	Feb 2001	7	4A	Aug-Sept 2000	0	0/1
Keperra	14/4/2000	Feb 2001	10	4A	May-Jun 2000	0	0/1
Grange	24/10/2000	Feb 2001	4	A2E	July-Aug 2000	0	0/1
Grange	21/12/1999 24/10/2000	Feb 2001	14 4	A2E A4E	Aug-Sept 1999 Aug Sept 1999	0	1/2
Grange	10/5/2000	Feb 2001	9	4A	Jun-Jul 2000	0	0/1
Grange	21/12/1999	Feb 2001	14	3A	Apr-May 2000	0	0/1
Grange	24/10/2000	Feb 2001	4	4A	Nov-Dec 2000	0	0/0

Table 3 continued

Grange	21/12/1999 8/5/2000 24/10/2000	Feb 2001	14 9.5 4	A A2 2A	Dec 1999 Dec 1999 Jan 2001	1	1/1
Site	Date tagged	Final recapture date	Time at large months	Position of UV mark	Estimated time of formation of check nearest to OTC mark	Additional checks post first OTC mark	Expected additional checks if formed winter-spring/summer-autumn
Grange	10/5/2000 24/10/2000	Feb 2001	9.5 4	4A	Jun-Jul 2000	0	0/1
Grange	21/12/1999	Feb 2001	14	A2	June-Aug 1999	1	1/2
Grange	23/10/2000	Feb 2001	4	3A	Dec-Jan 2000	0	0/0
Grange	21/12/1999 8/5/2000	Feb 2000	14 9.5	4A 1A (same ann.)	Feb 2000 Feb 2000	1 (est. summer)	1/1
Tabragalba	13/4/2000 2/11/2000	Feb 2001	10.5 4	A2E A3E	Dec-Jan 99/00 Dec-Jan 99/00	0	1/1
Tabragalba	7/7/1999 5/10/2000	Feb 2001	19 5	4A A4E	Aug-Sept 1999 Aug-Sept 1999	0	1/2
Tabragalba	7/7/1999	April 2000	9	A1E	May-Jun 1999	0	0/1
Tabragalba	2/9/1999	April 2000	7	A	Sept 1999	0	0/1
Tabragalba	2/9/1999	April 2000	7	3A	Oct-Nov 1999	0	0/1
Tabragalba	2/9/1999	April 2000	7	A3E	May-Jun 1999	0	0/1
Samford	12/10/2000	Feb 2001	4.5	A3E	Jun-Jul 2000	0	0/1
Samford	29/7/2000	Feb 2001	7	4A	May-Jun 2000	1	0/1
Samford	13/10/2000	Feb 2001	4	A4E	May-Jun 2000	0	0/1
Samford	30/7/1999	Feb 2001	19	2A	Sept-Oct	1	1/2
Samford	29/7/2000 12/10/2000	Feb 2001	7 4	A A2E	Jul 2000 Sept 2000	1 (spring)	0/1

Table 4: Summary of ageing validation data for OTC marked wild caught eels (n = 35).

	Estimated time of formation of check nearest to first OTC mark		Estimated periodicity of check formation		
	Winter–spring	Summer–autumn	Annual	Greater than annual	Less than annual
Count	29	6	29	3	3
Per cent	82.86	17.14	82.86	8.57	8.57

Relationship between otolith growth and somatic growth

Figures 4 to 8 show the relationship between otolith growth (growth to edge and growth to OTC mark) and somatic growth in wild eels and tank held eels. Growth to the OTC mark, represents only a short period held in tank conditions for tank held eels, whilst growth to the edge of the otolith represents a period in captivity of at least 18 months for tank held eels. The regression equations for the relationship between somatic growth and growth to the OTC mark or growth to the edge of the otolith are similar in wild eels (Figure 5). There is no significant difference between slopes ($p = 0.579$) or intercepts ($p = 0.229$). The regression for wild eels total length versus distance to the edge of the otolith were also similar to the regression of somatic growth and growth to the OTC mark in Tank held eels (Figure 6) with no significant difference in slopes ($p = 0.616$) or intercepts ($p = 0.538$). Similarly there were no significant differences in the regressions of somatic growth versus distance to the OTC mark in wild eels or tank eels (slope $p = 0.973$, intercept $p = 0.654$). Thus it was legitimate to pool the OTC data for tank and wild eels (Figure 7). As would be expected, the pooled OTC data regression equation was also similar to the regression equation for somatic growth versus distance to the edge of the otolith in wild eels, with no significant difference in slopes ($p = 0.523$) and intercepts ($p = 0.347$).

In contrast, the regression equations for the relationship between somatic growth and growth to the edge of the otolith for tank held eels are unlike the previous examples listed above. The slope of this regression is not significantly different from the slope of the regression ($p = 0.147$) for wild eels for growth to the edge of the otolith, but the intercepts of the two regressions are significantly different (<0.001). Both the slope ($p = 0.05$) and intercept ($p <0.001$) of the regression for somatic growth against otolith growth to the OTC mark are significantly different to those of the regression for somatic growth against growth to the edge of the otolith for tank held eels (Figures 6 and 8). Similarly the regression for distance to the OTC mark versus somatic growth in wild eels was different to the regression for distance to the edge of the otolith versus somatic growth in tank eels (slope, $p = 0.037$ intercepts $p <0.001$). The pooled distance to OTC mark-somatic growth relationship was also significantly different to the otolith edge-somatic growth relationship in tank held eels (slope $p = 0.018$, intercepts $p <0.001$).

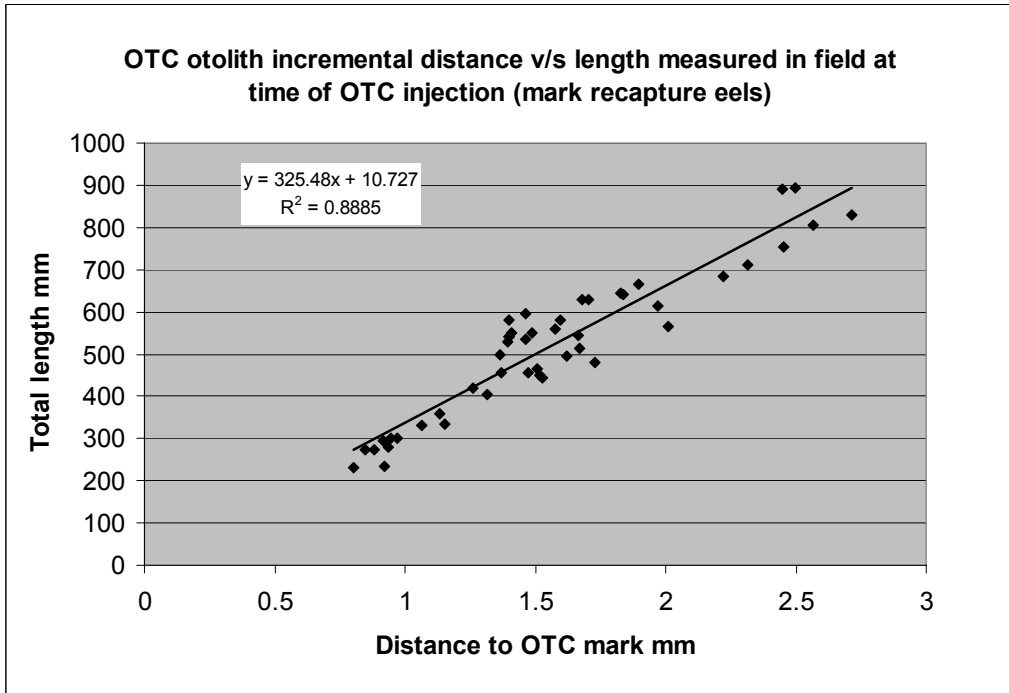


Figure 4: Plot of the relationship between somatic growth of wild longfinned eels and growth of otoliths to the OTC mark. The linear regression equation and R-squared value are displayed on the plot.

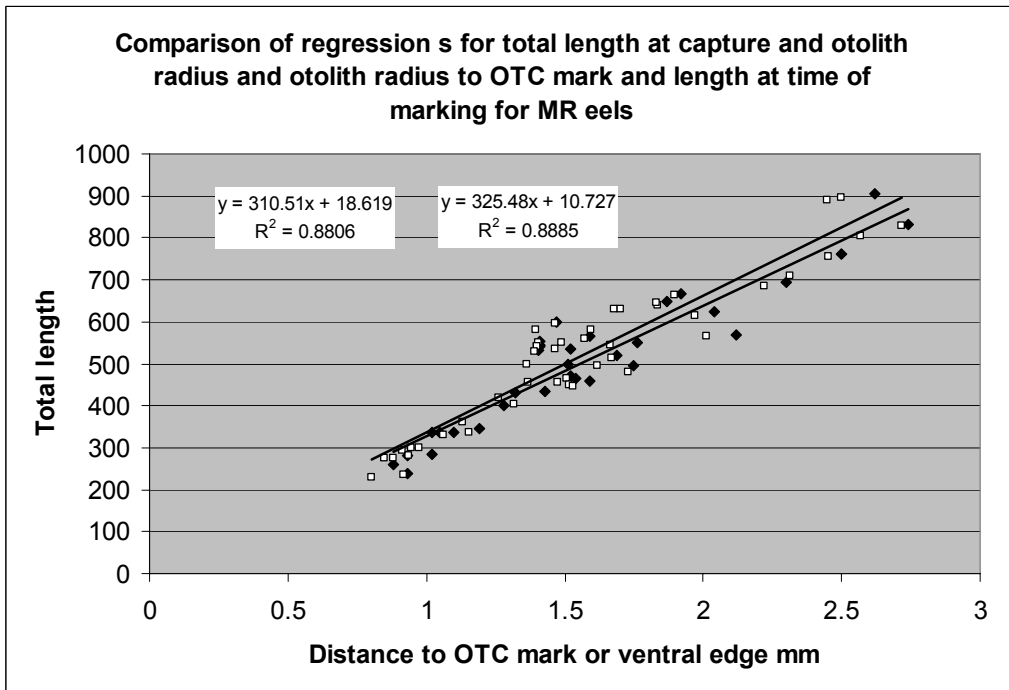


Figure 5: Plot of the relationship between somatic growth of wild longfinned eels and growth of otoliths measured to the edge (black diamonds), compared with plot of the relationship between somatic growth of wild longfinned eels and growth of otoliths to the OTC mark (white squares). The linear regression equation and R-squared values for each are displayed on the plot with the equation for growth of otoliths to the edge on the left.

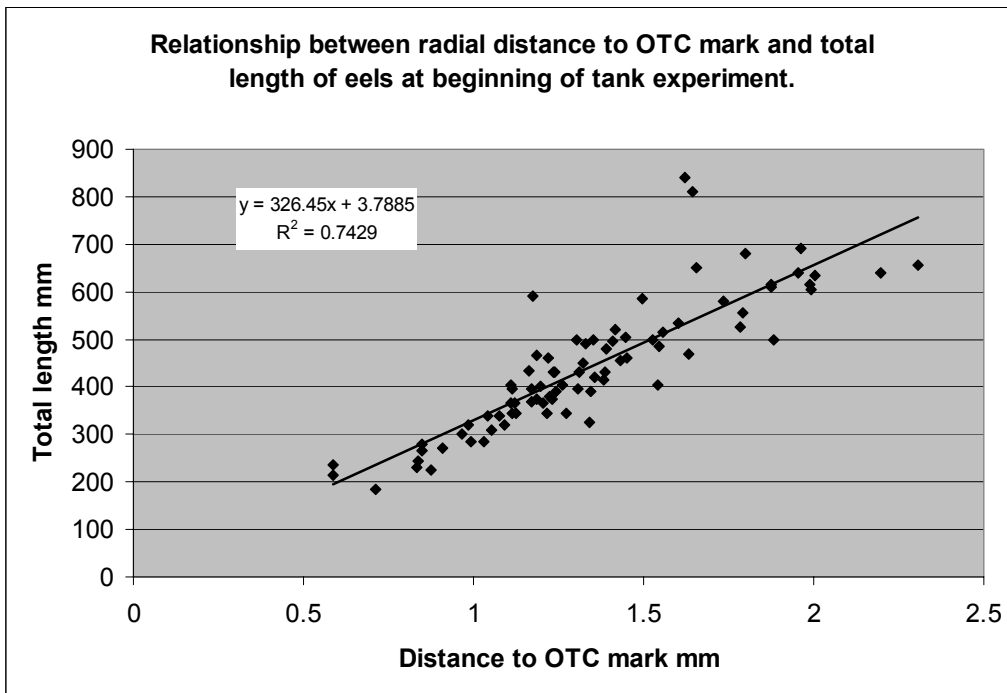


Figure 6: Plot of the relationship between somatic growth of longfinned eels and growth of otoliths to the OTC mark. Eels were held in tanks for approximately two months prior to OTC injection. The linear regression equation and R-squared value are displayed on the plot.

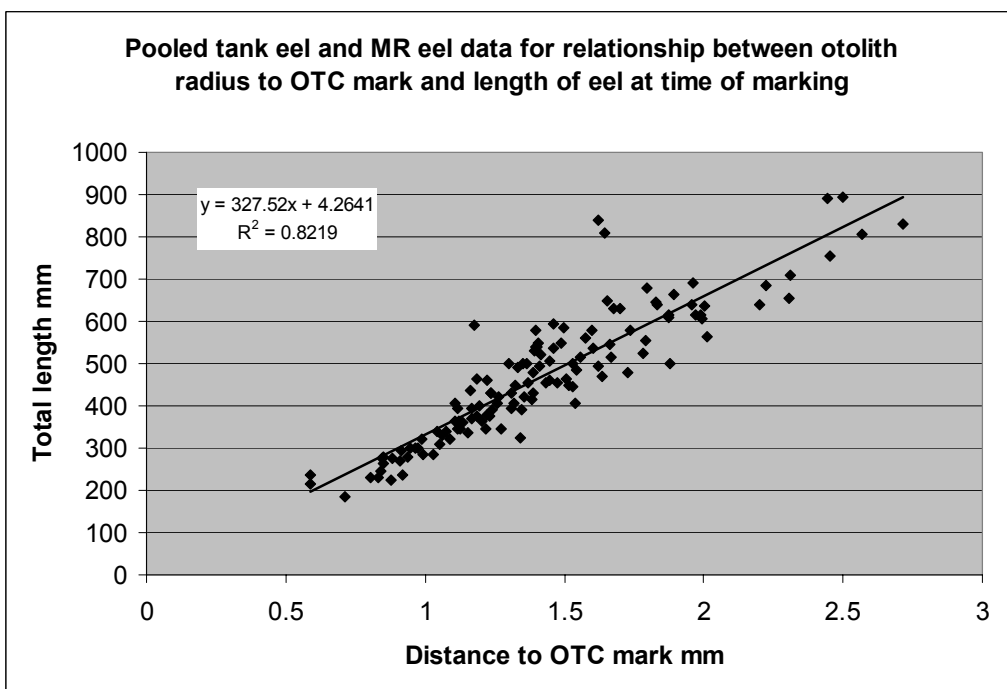


Figure 7: Plot of the relationship between somatic growth of longfinned eels and growth of otoliths to the OTC mark using pooled data for wild marked-recaptured eels and eels OTC tagged after being held in tanks for less than two months. The linear regression equation and R-squared value are displayed on the plot.

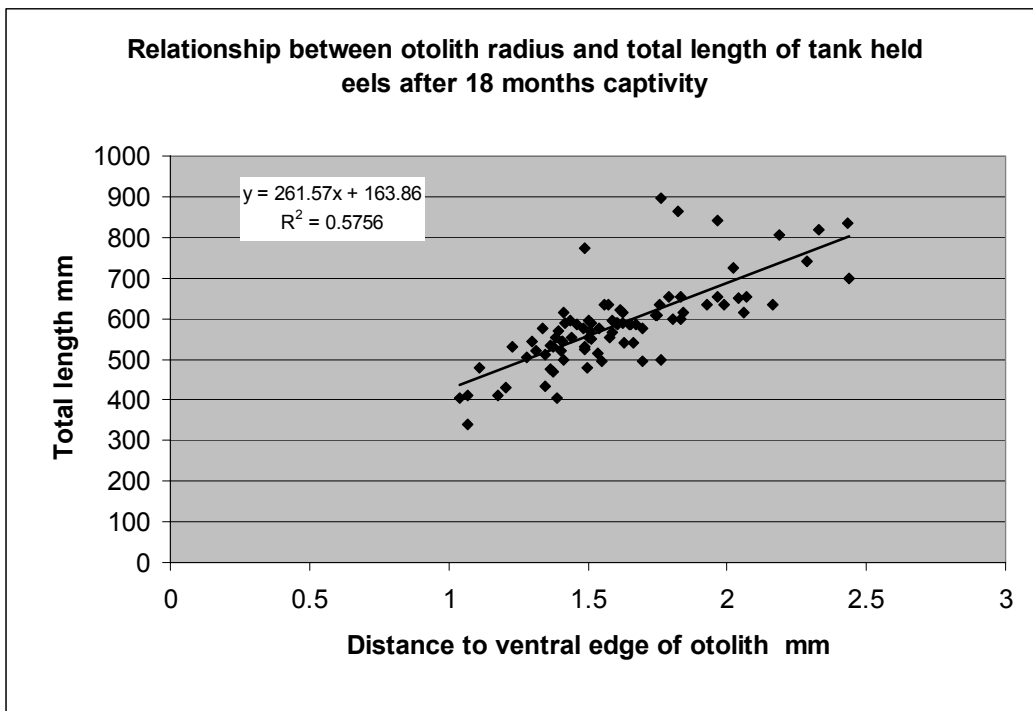


Figure 8: Plot of the relationship between somatic growth of eels held in tanks for 18 months and growth of otoliths measured to the ventral edge. The linear regression equation and R-squared values are displayed on the plot.

Discussion

Timing and periodicity of check formation

Tank eels

Timing of check formation in the tank held eels was predominantly winter-spring, but with a small number of eels forming checks as early as March. It is possible that check formation could have been in response to handling stresses at the time of OTC injection leading to correspondence of check formation with winter. However, this is unlikely for the majority, as eels were handled quarterly for weighing and measurement and few eels (<3 per cent) formed greater than the expected number of checks post-OTC injection.

If checks were forming annually, then only one additional check would form following formation of the check closest to the OTC mark. This was the case for more than 85 per cent of the eels. Nevertheless, just over 12 per cent of eels had no additional check form in the 18 month period following OTC injection. Of those eels that failed to form additional checks, eight of the ten were between 350 mm and 500 mm at the time of OTC tagging. The mean increase in length for these eels was 11.34 per cent (± 8.41 per cent) by the end of the experimental period. In eels that did add check marks from the same size group, the mean increase of body length was 36.70 per cent (± 14.96 per cent) over the same period. Therefore it appears that it is the slower growing, possibly socially dominated eels that were not forming check marks. If these eels were socially dominated, then their growth may have been consistently slow over summer and winter. Slow growth alone would not seem to preclude check formation as there were some individual eels in the tanks with similar slow growth rates that did form checks. Nevertheless the majority of eels that formed checks had faster growth than those that did not. It is likely that variable growth between summer and winter was the key to check formation. The other two eels with no check mark formation had

body lengths of 640 mm and 840 mm at the time of tagging. They increased in body length by 15.62 per cent and 6.55 per cent respectively over the period of the experiment. This is within the range of growth of the 350 mm–500 mm size class of eels that did not form checks.

Ishikawa *et al.* (2000) found no distinct rings forming in aquacultured *A. japonica* held in captivity for two to five years from the glass eel stage. This occurred in both eels held under constant temperature conditions and in eels held in conditions where the temperature fluctuated with the seasons between 16°C and 26°C. Slow growth would not appear to be the cause of no check formation in these Japanese eels. It is more probable that the levels of food available under the cultured conditions were greater than those for wild eels in winter. The resulting fast or unchecked growth rates in the Japanese eels most likely prevented formation of check marks. In contrast, in our experiment feed rates were reduced in the winter period in correspondence with reduced eel activity and lower temperatures. Growth rates of most eels held by us were rapid over the warmer months and probably greater than those of most wild eels.

Overall the results of the experiments for tank held eels supports the hypothesis of annual formation of check marks and formation of checks in winter-spring.

Wild eels

Validation of annual check formation in tank held eels is by itself not sufficient to confirm that wild eels form check marks annually. The data from the wild eels is therefore important to support observations from tank held eels. As for eels from NSW (Pease *et al.*, 2003), the timing of annulus formation in Queensland eels was highly variable, but the majority of checks formed in winter and spring (83 per cent). However, some eels formed checks in the summer-autumn period (17 per cent). In more temperate regions eels form checks only during winter (Oliveira, 1996; Chisnall and Kalish, 1993), but supernumerary or partial checks may form in summer. In this study eels with summer check formation were captured in the same locations as eels with winter-spring check formation. There is no doubt that summer could create stressful conditions which could lead to complete ring formation in some individuals. It is also possible that some of the summer rings observed may have been due to handling stress associated with capture and OTC injection. This was observed by Oliveira (1996) for some *A. rostrata* after handling. Despite the possibility of handling induced complete rings in summer, some individual eels formed summer marks that could not have been related to handling (Table 3) and one of these eels formed two summer checks without an intervening winter check mark.

The majority of wild eels in this study had checks forming at a rate consistent with annual formation (83 per cent) but 8.5 per cent had complete rings form at a rate greater than annual. Additional complete rings could have been related to handling stress (at least some marks corresponded closely to OTC injections at recapture) or to check formation in both summer and winter in some individuals. Nevertheless an equal proportion of eels formed checks less than annually. These latter individuals all had poor growth rates, increasing their body length between 2.7 per cent and 4.3 per cent over periods ranging from 10.5 to 19 months. This result is consistent with the result for eels that did not form checks under tank conditions.

Based on the results of the validation experiments for wild eels and tank eels it would appear that the majority of *A. reinhardtii* in subtropical Queensland form complete check marks annually. Although some individuals form check marks at greater or lesser intervals, these occur at a sufficiently low rate and balance each other out so as not to bias age-based population models. Therefore age-based modelling on the assumption of annual check formation is valid.

Relationship between otolith length and eel total length

The regression equations for total length of eels at time of tagging and distance to the OTC mark on the otolith was similar for both tank held eels and wild eels. The same was true for the relationship between eel total length and distance to the edge of the otolith in wild eels when compared with eel length at time of tagging versus distance to the OTC mark. R-squared values for all these regressions ranged from 0.74 to 0.89 and are within the range reported for numerous other fish species (Harvey *et al.*, 2000). In contrast to the above relationships, the relationship between total eel length and distance to the edge of otoliths in tank held eels had a much lower R-squared value (0.58) and a significantly different intercept to all the other regressions. The slope of this regression was also significantly different to all the regressions involving distance to OTC mark. The relationship between somatic growth and otolith growth appears to have changed or broken down following captivity. Francis *et al.* (1993) refer to the uncoupling of otolith and somatic growth in snapper (*Pagrus auratus*). They noted that otoliths in slower growing snapper were comparatively larger than otoliths of fast growing snapper of the same length. This is because otoliths continue to grow, even when somatic growth has slowed or stopped (Campana, 1990). It is likely that most of the eels held in tanks had growth rates substantially higher than most of the wild eels and that increased somatic growth rates outstripped any increase in otolith growth rates. Growth of otoliths to the OTC mark in tank held eels represents only a small period in captivity (less than three months), therefore sufficient time had probably not elapsed for the breakdown in the somatic–otolith growth relationship to be detected as compared to eels after another 18 months in captivity.

The strong linear relationships and high R-squared values derived from otolith and somatic growth data from wild eels confirms proportionality in otolith growth and somatic growth. Therefore there is potential for back calculation of eel size at age based on otolith measurements. Nevertheless the derived linear relationships are undoubtedly composed of eels that had a range of growth rates. Given that slower growing fish will have comparatively larger otoliths than faster growing fish of the same size (Campana, 1990) then the regression trajectories of individual fish otoliths will differ from the regression trajectory calculated for the whole population. Through using a biological intercept (i.e. one that is biologically and not statistically based) it should be possible to accurately calculate the regression trajectories for individual eels and therefore back calculate growth (Campana, 1990) provided there has not been variable growth through the life of the eel. Therefore it is recommended that any back calculation equations should not be based on data from captive eels as our data has demonstrated how uncoupling of the otolith and somatic growth can occur in such a situation. Nevertheless, Campana (1990) also provides a method to correct for variable growth rates

Conclusions

- Complete check marks are formed annually in over 80 per cent of longfinned eels in sub-tropical Queensland.
- Timing of annulus formation is variable but the majority of annuli are formed in the winter spring period, with a small proportion being formed in summer.
- It is acceptable to use check marks from *A. reinhardtii* otoliths for age based modelling.
- The relationship between somatic growth and otolith growth is strongly linear in wild eels, therefore otoliths can be used for back calculation of length at age.
- There is likely to be uncoupling of otolith and somatic growth relationships in aquacultured eels or in populations with rapid growth rates.

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Chapter 5: Australian longfinned eel parameter estimation

Simon Hoyle and David Peel

Abstract

Female longfinned eel maturation was observed between the ages of 5 and 39 years, and male maturation between 4 and 17 years. Growth, survival, catchability and maturity parameters were estimated using a Bayesian equilibrium population dynamics model. Data for the model was derived from eels collected in both Queensland and NSW. Maturity parameters showed females maturing, as expected, larger than males, with a mean length of 955 mm against 530 mm for males. Annual probability of maturation was lower for females with median of 0.2 versus 0.93 for males, but this parameter was not well defined for females. There was no evidence of significant difference between male and female lengths at differentiation, in either Queensland or New South Wales. However, there was a very large difference in the values estimated for the two states. It appears that both sexes begin to differentiate at about 200 mm in Queensland, while differentiation does not begin until 400 mm in New South Wales. However, rather than evidencing an actual difference in length of differentiation between the areas, this difference reflects the fact that NSW eels came either from a commercial source or were caught using commercial fishing techniques. The NSW estimated length at differentiation is lower than the length at 50 per cent selectivity, suggesting that it is probably biased. Age differed between sites and increased with length. However, there was no evidence for differences between sexes.

In Queensland the ratio of eels differentiating into males versus females in the sampled area was estimated as 3.35, a significant bias towards males in the areas sampled. However, females live longer than males, so the sex ratio present in the population at any time would be more balanced.

Selectivity differed between Queensland and New South Wales, reflecting the different fishing methods. Length at 50 per cent catchability was 560 mm in New South Wales using commercial fishing techniques; and 175 mm in Queensland, using fishery-independent electrofishing.

The female silver eels obtained in this study had grown more slowly than males. This is different from the results of many other studies, where females have tended to grow faster than males. The far broader and more representative sampling of non-migrating eels found no differences between male and female age at length when location was taken into account. This suggests that any difference between sexes in growth rate may be environmental.

Project objective

Estimate population parameters required for a management model. These include survival, density, age structure, growth, age and size at maturity and at recruitment to the adult eel fishery. Estimate their variability among individuals in a range of habitats.

Introduction

The Queensland and New South Wales eel fisheries are predominantly for long-finned eels and respectively take 44–57 tonnes and 120–450 tonnes annually (Wohlsen *et al.*, 2002 and Pease, 2000). These fisheries are worth on average \$0.5m and \$1m per year respectively (QFS C Fish database and Pease, 2000). Glass eel harvesting supports aquaculture industries valued at \$117 400, \$433 000 and \$893 400 in 2001–02 in NSW, Victoria and Queensland respectively (ABARE). Adult eel fisheries also provide valuable employment in regional areas. Victorian eel fisheries are mostly for shortfinned eels *Anguilla australis*, with longfinned eels *A. reinhardtii* making up only five per cent of the 280 tonne average annual catch (McKinnon, 2002).

In Queensland the commercial adult eel fishery is a trap fishery and is restricted to freshwaters impounded by an artificial human made structure (e.g. a dam weir or barrage); a definition that includes farm dams (Zeller and Beumer, 1996). The fishery is further restricted by a prescribed number of traps, and by weekend closures (Fisheries (Freshwater) Management Plan, 1999 reprint No. 2B, 2003). There is no total allowable catch restriction due to the current small number (26) of licensed trappers (Wohlsen *et al.*, 2002).

The NSW eel fishery is also a trap fishery, restricted to estuarine and impounded waters. Non-tidal riverine areas are closed to commercial harvest (Pease, 2000). Approximately 200 eel fishers are endorsed to fish in estuaries, and about 13 fishers permitted to fish impoundments and farm dams.

Management of these fisheries has the objectives of ensuring sustainability, and maximising yield from both the adult and glass eel fisheries. Determining appropriate management measures requires an understanding of the likely effects of management changes — information best obtained through management strategy evaluation using population dynamics models.

In addition, the Australian federal government has introduced legislation requiring state governments to demonstrate that all export fisheries are being managed sustainably. Demonstrating this for eel fisheries will be facilitated by modelling of the fishery and the management regime.

Such models require estimates of population parameters: providing such estimates is the purpose of this study.

Estimating model parameters in a Bayesian context is particularly suitable for eel fisheries, and when the estimates are to be used in management strategy evaluation. Bayesian methods permit expert knowledge and data from similar populations to be incorporated into the model in the form of prior distributions. They allow multiple sources of information to be included in a single model, by combining likelihoods. They also permit effective estimation of uncertainty, maintain the correlations between parameters, and allow uncertainty to be correctly distributed through the model outputs. When applied to management strategy evaluation, the joint posterior distributions of Bayesian models provide appropriately distributed estimates of the likely outcomes of alternative management procedures, and the uncertainties in these outcomes.

As noted above, the Queensland and New South Wales eel fisheries are targeted mainly at the Australian longfinned eel, *Anguilla reinhardtii*. The shortfinned eel, *A. australis*, is also taken but not in significant quantities. The two species probably have similar growth rates in the same habitat, but their distribution, habitat usage and length at maturity are very different. These factors imply a need for different management techniques (Hoyle and Jellyman, 2003). Prior to this study there were no estimates of growth rate or age at maturity for *Anguilla reinhardtii* in Queensland or New South Wales, and therefore no models of longfin populations.

Eels internationally have been well studied, due to the high value of the various eel fisheries. Numerous studies have been made of growth rates and age and length at maturity. However, there are two persistent problems in estimating parameters for population models: growth variability, and parameter confounding.

Growth is highly variable in eels. Variation in growth rates occurs between habitats, and between individuals in the same habitat (Jellyman, 1997). Growth is affected by food supply, density, and temperature (Graynoth and Taylor, 2000; Graynoth and Taylor Submitted). Eel growth rate estimates can depend on the selectivity of the sampling technique, since growth estimated from length at age will be biased when not all lengths at each age are sampled, or if fishing mortality selectively removes some size classes. This problem can be resolved either by ensuring that all lengths at each age are sampled (e.g. Graynoth and Taylor Submitted), or as in this study, by including both selectivity and growth in a single model.

Confounding of parameters is in fact a persistent problem in many parameters. As with selectivity, and that fishing mortality may be confounded with growth rate. Emigration of mature eels appears to be a function of length to be introduced in fisheries (Vollestad, 1992), so growth rate estimates that do not take account of migration may be too low. Emigration of mature eels has the same effect on the catch curve as does mortality, so maturity and mortality parameters are confounded.

We used a Bayesian model of longfinned eel population dynamics to estimate the population parameters required for a management model. These parameters included survival, growth, size at maturity and size at recruitment to the adult eel fishery. We also estimated their variability among individuals in a range of habitats.

Methods

Data collection

Sampling in Queensland concentrated on freshwater reaches since the females tended to concentrate in these areas. Female population dynamics are more sensitive to fishing and more important to the sustainability of glass eel and adult fisheries. Data on estuarine and male population dynamics from New South Wales were complementary to the Queensland data. New South Wales data from freshwater reaches was used to investigate latitudinal variation in growth rates and age at maturity. New South Wales data were supplied by Bruce Pease and were collected as part of FRDC project 98/127.

Sampling was restricted to south-east Queensland, with Bundaberg the northern limit. Downstream migrations of silver eels were opportunistically collected from traps set in the Bundaberg Barrage fishway on the Burnett River. Sampling was stratified into reaches in small streams, reaches in large streams, and impounded waters. These included four sites on Wivenhoe dam, ten sites in other impoundments both previously fished and unfished, ten sites in small streams that had not been fished, ten sites in larger unfished waterways, and ten sites in larger fished waterways. Sites on two of the streams also had glass eel fishing at the mouth, and glass eel sampling occurring under FRDC project 97/312 (see Chapter 1 for details). At all sites both longfinned and shortfinned eels were sampled, but longfinned eels comprised almost all of the catch.

Within these sites, transects were chosen as randomly as practicable. Each waterway was divided into 10 by 10 km blocks, and three blocks selected at random. Within these blocks, all suitable access points were determined, and one of these was randomly selected as an appropriate starting point.

There was not sufficient statistical power to compare fished and unfished sites in impoundments, since catch rates were low in all impoundments sampled. This may have been caused by poor recruitment due to the high barriers downstream from impoundments, or by past documented and undocumented fishing. For each site distance from the sea, existence of barriers to migration such as barrages, depth and area of reach, and habitat type were recorded.

At each site the size structure of the population was investigated by electrofishing along one hundred-metre transects. In large streams each of three transects was electrofished twice — once on each side. On small streams six transects will be fished in series. The methods used for fishing impoundments depended on the size of the impoundment. Up to six transects were run in each impoundment. The length and sex of each eel obtained was recorded. Turbidity and water temperature were also recorded.

At each site all electrofished eels were taken for ageing. Sampled eels were aged in order to estimate age structure (and therefore mortality rates), growth rates, and their variability. We also examined the feasibility of estimating mortality rates of shortfinned glass eels in southeast Queensland, however, too few specimens were collected.

NSW Fisheries also provided length age and sex data for eels collected at estuarine sites and some freshwater locations in NSW. Details are contained in FRDC report 98/127.

Sampled eels were examined to determine sex ratios and reproductive status. Condition indices and length-weight relationships were determined. For sex determination, gonads were examined by eye, or by binocular microscope for individuals that were difficult to sex (Queensland) or below 600mm (NSW). In New South Wales all male and undifferentiated gonads were examined histologically.

Model components.

Growth, survival, catchability and maturity parameters were estimated using a Bayesian equilibrium population dynamics model. Sampling, using sampling importance resampling (SIR) was used to integrate out nuisance parameters and generate posterior distributions of model parameters.

Bayesian techniques are becoming the standard in fisheries modelling (e.g. (Punt and Hilborn, 1997; McAllister and Kirkwood, 1998; Meyer and Millar, 1999; Maunder and Starr, 2001), since they allow more straightforward integration across various sources of uncertainty, and permit unbiased prediction of management outputs. The joint posterior distribution of parameters includes both parameter uncertainty and covariance, which is very suitable for forward projection. They allow relevant information to be introduced from other stocks, fisheries, or expert opinion, via prior distributions, for example to restrict parameters to biologically plausible values. For an introduction to Bayesian methods in fisheries see (Punt and Hilborn, 1997) or (McAllister and Kirkwood, 1998).

Each eel was initially assigned a sex according to the sex ratio σ . Although eel sex is thought to be determined environmentally rather than genetically, this modelling strategy was appropriate since no environmental variation was included in the model.

Transition from the undifferentiated state into a state in which sex could be determined was

modelled as a function of length L and sex s , using the equation $D_{L,s} = \frac{\gamma d_s}{1 + \exp\left(\frac{\lambda d_s - L}{\eta d_s}\right)}$ (Equation

1), where γd is the maximum rate of differentiation, λd is a semi-saturation constant, and ηd is inversely proportional to the slope of the differentiation curve at $L = \lambda d$. This equation is based on the length-maturity relationship equation suggested by de Leo and Gatto (1995).

Similarly, the probability that an eel of length L and sex s will become mature (μ_L) was calculated

by the equation: $\mu_{L,s} = \frac{\gamma m_s}{1 + \exp\left(\frac{\lambda m_s - L}{\eta m_s}\right)}$ (Equation 1) (de Leo and Gatto 1995), where m is the

maximum rate of metamorphosis, λm is a semi-saturation constant, and ηm is inversely proportional to the slope of the metamorphosis curve at $L = \lambda m$.

The length L of an eel of age i was calculated by

$$L = (a_1 + b_1 i) \cdot e^{a_2 \varepsilon} \quad (\text{Equation 2}),$$

where the parameters (a_1, b_1) define mean length as a linear function of age, a_2 describes the lognormal deviation of length-at-age, and ε is a standard normal variate representing between-

individual variation in length at age. This differs from the formulation used by Francis and Jellyman (1999) and Hoyle and Jellyman (2002) in using lognormal rather than normal errors. The number of growth classes used in conjunction with the parameter ϵ was variable. Preliminary estimates were carried out with nine growth classes, but this was increased to 39 to produce final results. Linear growth is characteristic of New Zealand eels, and has been observed for other eel species (Jellyman, 1997, and references therein). It may be a general feature of eel species, given eels' lack of investment in annual spawning (Jellyman, 1995; Francis and Jellyman, 1999).

Weight was calculated by $W = \alpha L^\beta$.

Exploitation rate at length L E_L was calculated using (Equation 3):

$$E_L = a + b(L - L_{\text{MinLW}}) \quad \text{if minimum legal weight} < L < \text{maximum legal weight}$$

$$E_L = 0 \quad \text{otherwise,}$$

where a and b are the intercept and slope of the size selectivity function. The size selectivity of the fishery overall has not been estimated, so a flat selectivity curve was assumed. However, the sensitivity of this assumption was investigated (see below).

The cohort of eels was followed from the age of one, when both maturity and exploitation are minimal, to age 42, when insignificant numbers remain.

Demographic model

Given the number $N_{i,c,s}$ of yellow pre-reproductive eels at age i in growth class c and sex s , we compute, in the following order:

- 1) eel length $L_{i,c}$ by using eq. 2, eel weight $W_{i,c}$ at length $L_{i,c}$.
- 2) proportion of eels in the population that will differentiate as males is σ
- 3) number of eels moving from the undifferentiated into differentiated state

$$D_{i,s,c} = \delta_{L,s} U_{i,s,c}$$

number of eels remaining after differentiation

$$U_{i,s,c}^* = (1 - \delta_{L,s}) U_{i,s,c}$$

- 4) number of eels undergoing metamorphosis at age i in growth class c , of sex s

$$M_{i,s,c} = \mu_{L,s} D_{i,s,c}$$

number of eels remaining after metamorphosis

$$D_{i,s,c}^* = (1 - \mu_{L,s}) D_{i,s,c}$$

- 5) number of eels surviving natural mortality

$$D_{i,s,c}^+ = (1 - e^{-M}) D_{i,s,c}^*$$

$$U_{i,s,c}^+ = (1 - e^{-M}) U_{i,s,c}^*$$

- 6) number of eels harvested at age i in growth class c , given E_L , the exploitation rate at length:

$$Y_{i,s,c} = E_L (U_{i,s,c}^+ + D_{i,s,c}^+)$$

number of eels escaping fishing and making up the following year's yellow stock:

$$U_{i+1,s,c} = (1 - E_L) U_{i,s,c}^+$$

$$D_{i+1,s,c} = (1 - E_L) D_{i,s,c}^+$$

The overall yield was thus given by the contribution by both sexes and all age classes and growth classes of legal weight, namely

$$Y^{\#} = \sum Y_{i,s,c} \quad (i = 1, 2, \dots, i_{max}, s = 1, 2, c = 1, 2, \dots, c_{max})$$

while overall biomass was:

$$W^{\#} = \sum W_{i,c} Y_{i,s,c} \quad (i = 1, 2, \dots, i_{max}, s = 1, 2, c = 1, 2, \dots, c_{max})$$

Metamorphosing biomass was:

$$MB^{\#} = \sum W_{i,c} M_{i,s,c} \quad (i = 1, 2, \dots, i_{max}, s = 1, 2, c = 1, 2, \dots, c_{max})$$

Model fitting

The expected catch during fishery-independent sampling at age and length was fitted to the observed data using a maximum likelihood model.

Observed data were translated onto a three dimensional matrix of sex (female (1), male (2), undifferentiated (3)), age (1 to 42), and length class (50 categories from 0 to 1500 mm). Observed silver eels were translated onto a separate matrix, but using only two sex categories instead of three.

Expected numbers in each cell of the above matrix were calculated from the expected numbers at age and length class $U_{s,i,c}$, $D_{s,i,c}$ and $M_{s,i,c}$. The eels in growth class c at age i were distributed amongst all the length categories that the growth class contacted. This distribution was carried out using the cumulative density function of the log-normally distributed length at age. The lower limit of each growth at age i class intersected a category ($startpoint_{i,c}$), with each $startpoint_{i,c}$ the $endpoint_{i,c-1}$ of the previous growth class.

The objective function indicating model fit was calculated by comparing observed and expected age-length distributions, according to age, length, and sex class. The multinomial likelihood model

$$LL = \sum_{s=1}^3 \sum_{i=1}^{i_{max}} \sum_{c=1}^{c_{max}} [x_{s,i,c} Ln(p_{s,i,c})]$$

used was

Parameters were estimated using a Bayesian statistical technique known as sampling-importance-resampling (SIR) (Macalister and Ianelli, 1997), described below.

Model selection was carried out using an ad-hoc version of the Bayesian information criterion

$$(BIC) \text{ (Congdon, 2001) page 474. } BIC' = \bar{L} - \frac{P}{2} \log n .$$

The model was implemented in MATLAB 6.5, using the Condor distributed computing system (Thain *et al.*, 2003) over 20 desktop machines to provide the large amount of computing time required.

Prior and proposal distributions

Bayesian estimation requires specification of prior distributions for all estimated parameters in the model. Values for which there is no prior information are assigned 'non-informative' priors. The starting length parameter, $a1$, was set to 52.0 ± 1.5 mm based on a published value for *A. reinhardtii* (Shiao *et al.*, 2002).

Prior distributions for all parameters are given in Table 1.

Table 1: Prior distributions for model parameters.

State	Parameter	Distribution	Parameter values	
			p1	p2
Both	a1	Normal	52	0.15
Qld	M	Logit Normal	-3	1.2
NSW	M	Logit Normal	-6	1.2

Proposal distributions

The proposal distributions, also known as the importance function (Berger, 1985) were chosen to maximise the efficiency of the algorithm. Sampling is more efficient when the proposal distribution matches the posterior distribution as closely as possible. Proposal distributions should 1) be easy to generate draws from, 2) have tails at least as dense as the tails of the posterior, and 3) mimic the true posterior density reasonably well (Oh and Berger, 1992) in (McAllister and Ianelli, 1997). Proposal distributions for all parameters were selected based on a series of trials with different values.

Estimating posterior distributions

Posterior distributions were obtained using the Sampling Importance Resampling (SIR) algorithm described in (McAllister *et al.*, 1994). The following procedure was followed:

1. Assign values to all parameters using samples from their proposal distributions. Record the proposal likelihood (probability of proposing the selected parameter values).
2. Project the population forward through N year classes for each of the c growth rate classes, recording the values of all parameters for which posteriors are required. Record the likelihood.
3. Repeat the above process for 400 000 iterations.
4. Use a combination of the likelihoods and the proposal likelihoods to assign relative probabilities to the results of each iteration.
5. Take 10 000 'resamples' from the iterations with replacement, with the probability of resampling each iteration proportional to its assigned relative probability.
6. The distribution of the above resamples is the posterior distribution of the parameters.

The various samples were separated into Queensland and New South Wales groups for the integrated Bayesian analysis. Within each group all samples were pooled, apart from the specifically-targeted silver eel samples. Thus growth rates in the New South Wales analyses included both estuarine and freshwater eels, but were dominated by the fishery-sourced estuarine samples. This method includes variation both between and within sampling locations in the growth rate variability parameter. Distributions for silver eel maturation parameters were estimated using the specifically-targeted silver eel data, and these distributions were used as prior distributions in all subsequent analyses.

Comparing growth rates

Growth rates were compared between sexes and locations, using general linear modelling of age at length. Age at length was used in preference to length at age, because selectivity and maturation affect fish at length rather than at age. When length at age is used, growth rate estimates are biased due to, for example, early loss of fast-growing males. Also, length is known quite accurately but age is uncertain. Only sexually differentiated eels between 350 mm and 700 mm were included, to avoid biases due to uncertain sex. Age at length was analysed using a general linear model with identity link function, length as a covariate, and location and sex as categorical variables. The analysis was carried out separately for Queensland and New South Wales data. Thus this analysis separated growth rate variation between sampling locations from variation within locations, by state.

Results

Figure 1 shows the annual probability of differentiation versus length (mm) for Queensland and New South Wales, and annual probability of maturation versus length for Queensland. Figures 2 and 3 show posterior estimates of female and male maturity parameters as estimated from silver eels from the Burnett River and Figures 4 to 6 show posterior distributions of various population parameters for Queensland and NSW.

Maturity and sexual differentiation

Maturity parameters estimated for Burnett River silver eels showed females becoming susceptible to maturation, as expected, larger than males (Table 2), with mean for the parameter λ of 760 mm against 533 mm for males. Annual probability of maturation was lower for females with median of 0.2 versus 0.93 for males, reflecting the far greater range of lengths at maturity for females. Distributions of these parameters are shown in Figure 1. For comparison, note that female maturation was observed between the ages of 5 and 39 years, and male maturation between 4 and 17 years.

Table 2: Posterior estimates for maturity, growth rate, and mortality parameters, for silver eels collected in fishway sampling at Burnett River Barrage. The maturity posteriors were used as prior distributions for maturity in the fishery-independent electrofishing sample analyses.

Sex	Parameter	Distribution	Estimated values		Posterior percentiles		
			p1	p2	2.5	50	97.5
Female	Mat γ	Beta	20.45	80.71	0.13	0.20	0.29
	mat λ	Normal	759.74	16.89	726.64	759.74	792.84
	mat η	Gamma	0.98	12.02	0.28	8.06	43.73
Male	mat γ	Beta	11.80	1.21	0.71	0.93	1.00
	mat λ	Normal	532.72	8.41	516.23	532.72	549.20
	mat η	Gamma	5.95	2.11	4.58	11.85	24.45

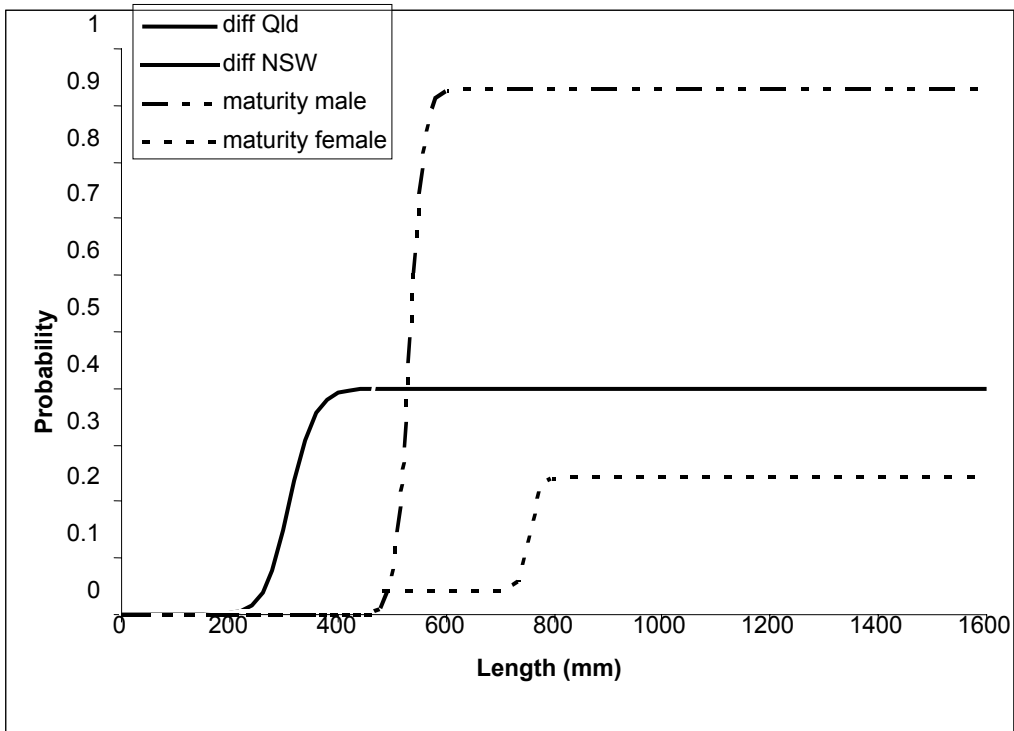


Figure 1: Annual probability of differentiation versus length (mm) for Queensland and New South Wales, and annual probability of maturation versus length for Queensland.

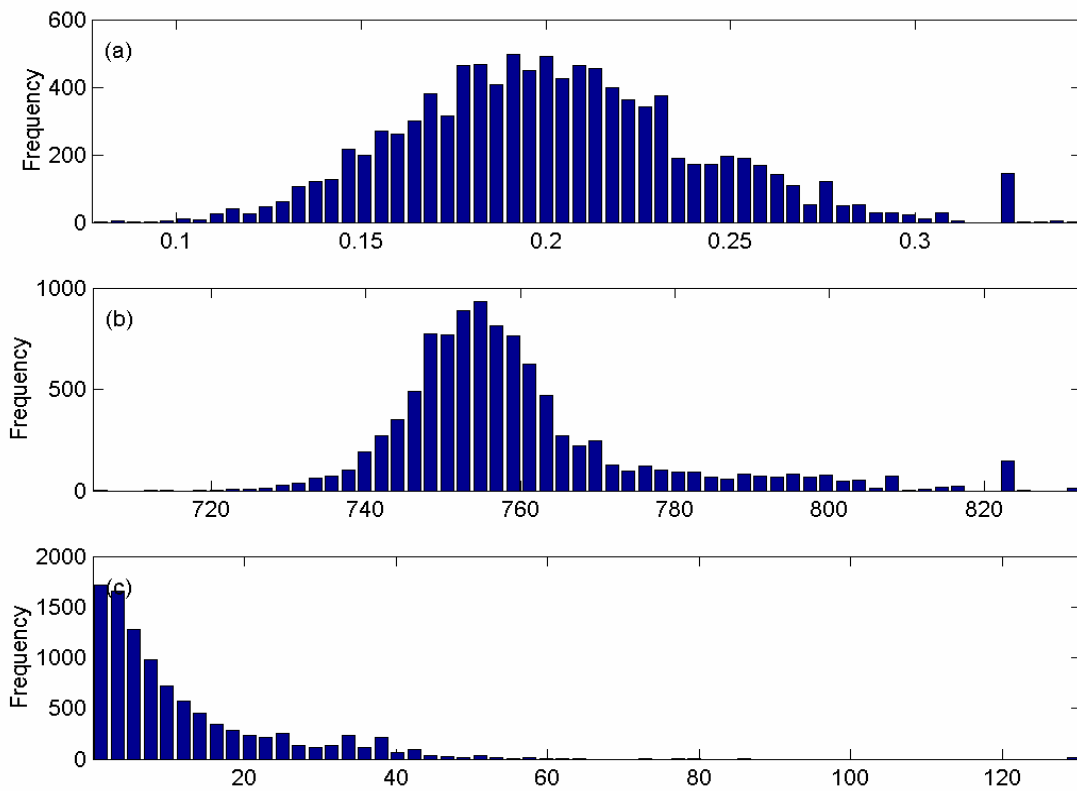


Figure 2: Posterior estimates for female maturity parameters calculated from Burnett River eels: (a) maturity ©, (b) L, (c) L. Parameters are defined in Table 1.

New South Wales maturation parameters (Figure 6) appeared somewhat different at first glance from the Queensland results, but in fact the differences were not statistically significant.

There was no evidence of significant difference between male and female lengths at differentiation, in either Queensland (Figure 4) or New South Wales (Figure 5). However, there was a very large difference in the values estimated for the two states. Differentiation was estimated as beginning between approximately 310 mm in Queensland, but not until about 480 mm in New South Wales (Figure 1). However, rather than evidencing an actual difference in length of differentiation between the areas, it is more likely to reflect different collection and analysis methods. Most NSW eels came either from a commercial source or were caught using commercial fishing techniques, and the 17 that were caught by electrofishing were generally large (average length 746 mm). The NSW differentiation lambda parameter (487 mm) is considerably lower than the length at 50 per cent selectivity (562 mm), which may have affected the result. If more eels had been caught smaller than 487 mm, a lower value may have been estimated. The much smaller Queensland length at differentiation is significantly higher than the local length at 50 per cent selectivity, so is unaffected by this type of bias. However, use of a macroscopic gonad examination method for the Queensland data, which is less accurate than microscopic examination in determining stage (Walsh et al, 2003, may have added to uncertainty in sex determination and caused bias. Different protocols can be reflected in the estimated length at differentiation.

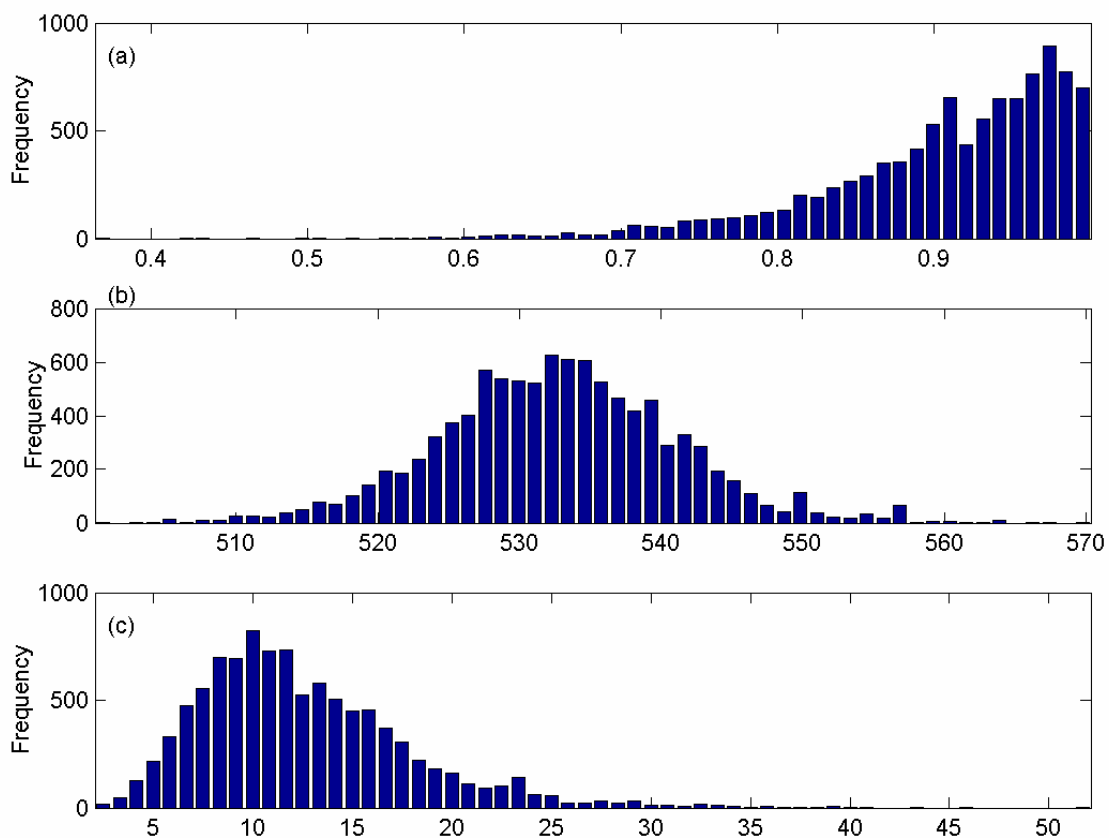


Figure 3: Posterior estimates for male maturity parameters calculated from Burnett Creek eels: (a) maturity λ , (b) L , (c) λ . Parameters are defined in Table 1.

General linear modelling of age at length

Age differed between sites ($p < 0.001$) and increased with length ($p < 0.001$). However, there was no evidence for differences between sexes ($p = 0.162$), or for the linear relationship with length differing between sites, since the length site interaction term was not statistically significant ($p = 0.421$). Parameter estimates are reported in Table 5.

$$Age = \alpha + \beta.length + \gamma_{site} + \varepsilon$$

NSW

The same analysis carried for data sourced from New South Wales gave similar results. Age differed between sites ($p < 0.001$) and increased with length ($p < 0.001$), but there was no evidence for differences between sexes ($p = 0.170$) or for the linear relationship with length varying between sites ($p = 0.302$). Parameter estimates are reported in Table 6.

Growth and mortality parameters

Estimates for the growth rate parameter $b1$ — the gradient of the growth rate — were higher for New South Wales than Queensland, possibly related to higher growth in estuarine habitats. There was more variability in the Queensland growth rates, with a higher estimate of the $a2$ parameter. Reflecting the range of habitats (small stream, large river, impoundment) and catchments sampled.

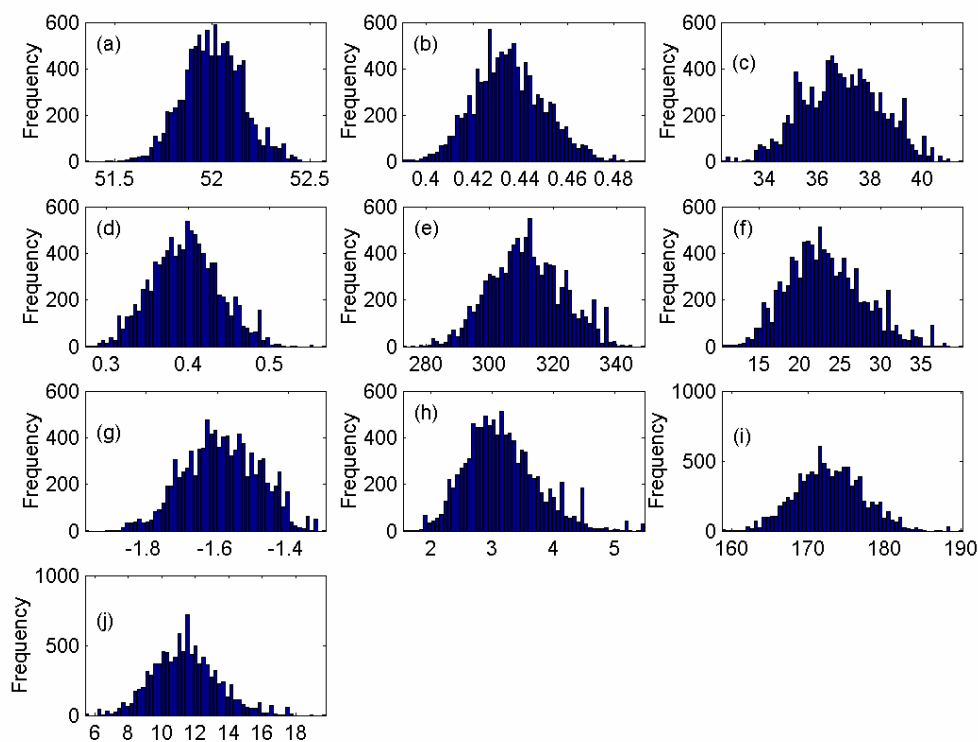


Figure 4: The posterior distributions of selected parameters for Queensland: (a) $a1$, (b) $a2$, (c) $b1$, (d) diff gamma, (e) diff lamda, (f) diff nu, (g) total mortality, (h) sex ratio, (i) mean selectivity, (j) selectivity slope. Parameters are defined in Table 1.

Other parameters

Sex ratio was not estimated for the New South Wales data, since the high limit of selectivity left few males in the population available for sampling. In Queensland the ratio of eels differentiating into males versus females in the sampled area was estimated as 3.35 ± 0.63 , a significant bias towards males in the areas sampled. However, females live longer than males, so the sex ratio present in the population had more females than the ratio at differentiation.

Selectivity differed between Queensland and New South Wales, reflecting the different fishing methods and habitats sampled. Length at 50 per cent catchability was 173 mm in Queensland, using backpack and boat-based electrofishing in riffles, pools and reaches; and 562 mm in New South Wales using commercial fishing techniques and boat-based electrofishing in pools and reaches. The New South Wales selectivity estimate is intermediate between a higher value in freshwater reaches than estuarine habitats also reflected in the higher value for selectivity slope in NSW (37.0 mm) than in Queensland (11.4 mm).

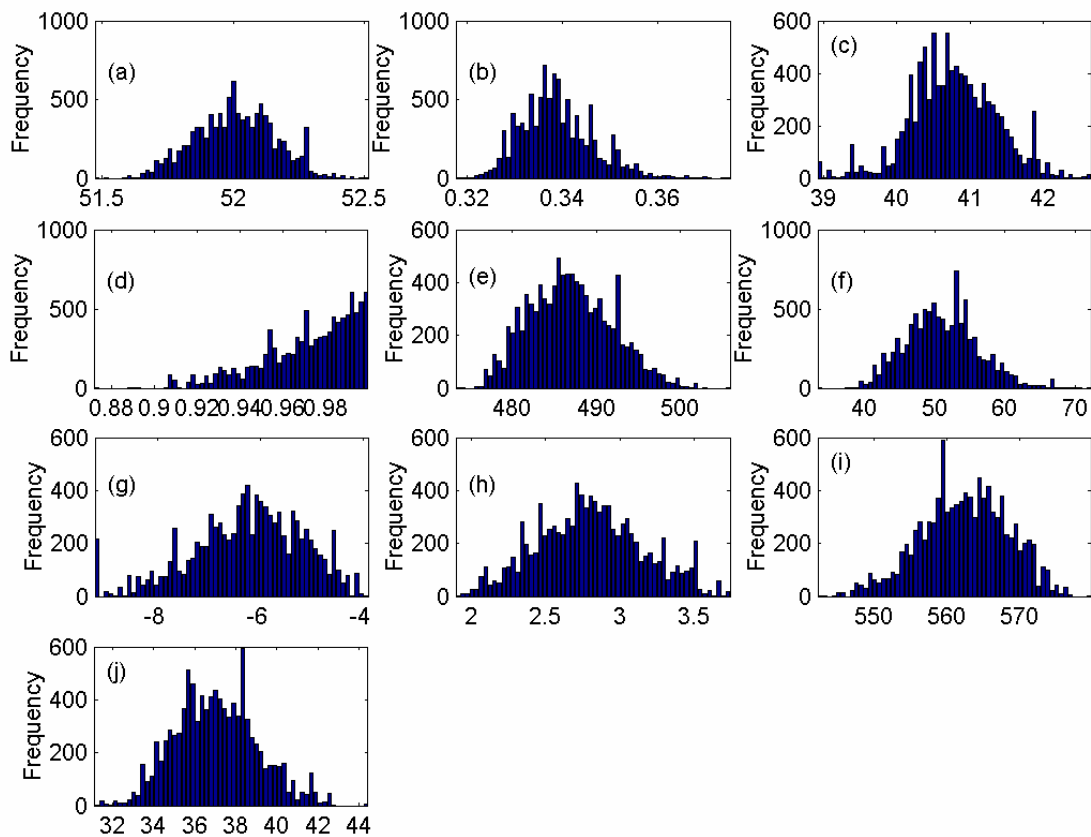


Figure 5: Posterior distributions for New South Wales growth and sexual differentiation: a) a_2 , b) b_1 , c) $\text{diff } \gamma$, d) $\text{diff } \lambda$, e) $\text{diff } \nu$, f) total mortality, g) mean selectivity, h) selectivity slope.

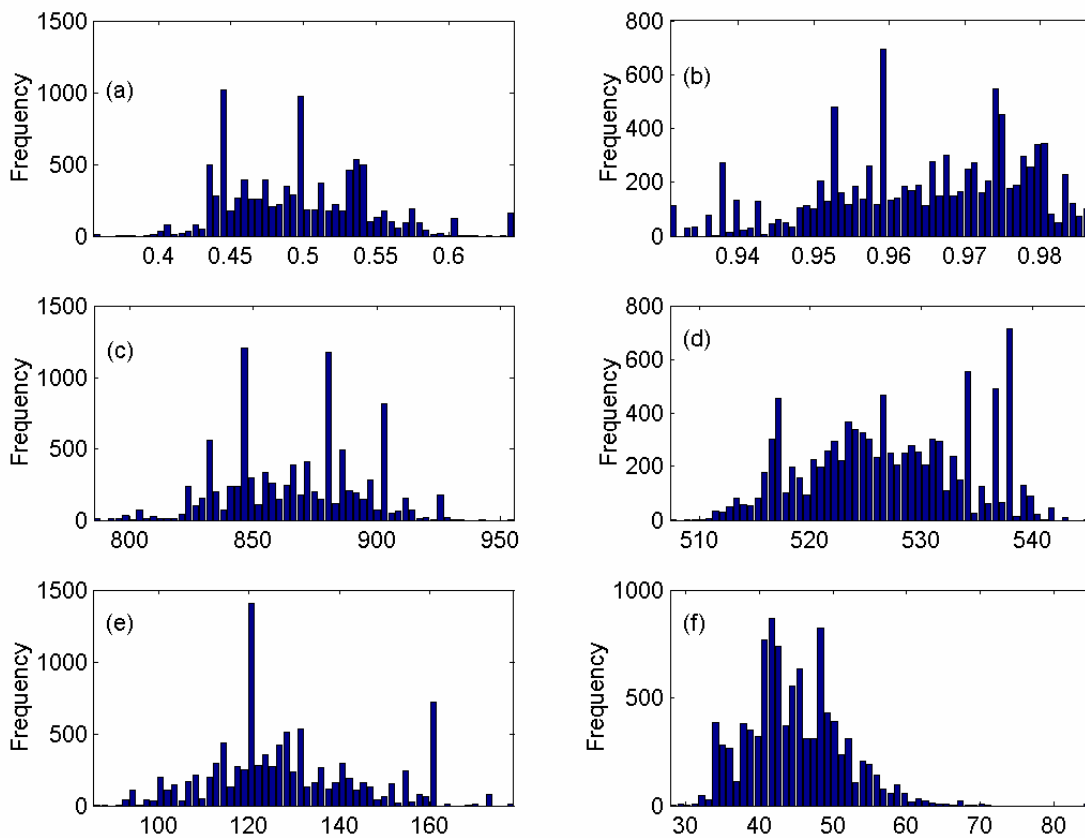


Figure 6: The posterior distributions of maturity parameters for New South Wales: (a and b) female and male maturity ©, (c and d) female and male maturity λ, (e and f) female and male maturity η. Parameters are defined in Table 1.

Discussion

The maturation parameters of male and female silver eels varied as expected, with the maturation parameter λ much larger for females than males, reflecting silver females maturing at larger size than males. Maximum annual probability of maturation (γ) was considerably lower for females than males, and the maturation parameter η was much larger for females than males, reflecting females' greater individual variation in length at maturity.

Males vs females

Differences between growth rates of male and female silver eels suggest that they may have developed in different parts of the Burnett catchment. These eels were sampled passing through the Ben Anderson Weir fishway, and may have come from any part of the Burnett catchment. These differing growth rates contrast with eels sampled in a single location, where growth rates did not differ significantly between sexes. These facts are consistent with the hypothesis that growth rate is driven by environmental conditions, and that males and females tend to occupy areas with different conditions. Sexes within an area may differ in growth rate, however, if their different sizes cause them to experience different conditions: e.g. females switching to piscivory at larger sizes (Jellyman, 2001).

The female silver eels obtained in this study had grown more slowly than males. In many other studies females have tended to grow faster than males (Poole and Reynolds, 1996) (Panfili *et al.*, 1994) (Oliveira 1999) (Helfman and Bozeman, 1984) (Vollestad and Jonsson, 1986) (Jellyman 1997). Few previous studies have found males growing faster than females (Holmgren *et al.*, 1997). This difference may simply have been due to our small sample size — one location on one day — although the single sampling location drains a large river and the flood conditions at the time led to many eels emigrating.

The far broader and more representative sampling of non-migrating eels found no differences between male and female age at length when location was taken into account — the same result occurred in both Queensland and New South Wales. This suggests that any difference between sexes in growth rate may be environmental.

Locations

Differences between male and female growth rates are likely to be driven largely by local landscape features. Growth rate is primarily driven by food availability (Graynoth and Taylor, 2000), and local density appears to drive sex determination (Krueger and Oliveira, 1999) but see (Holmgren, 1996) (Beullens *et al.*, 1997) (Oliveira and McCleave, 2000). Females tend to be found further upstream, where lower densities may lead to higher growth rates. Greater food availability in estuaries may support higher densities in upper tidal areas near freshwater (where juveniles may spend time before migrating further upstream (Walsh and Pease, 2004 and references therein). However, other factors may be confounded with differences in density, such as differences in the productivity of the landscape or habitat (estuarine versus freshwater). For example, in New Zealand, eel growth rates are faster in landscapes under pasture than in forest (Chisnall and Hicks, 1993), and rapid in low-density lakes (Beentjes, 2003). Another factor affecting apparent growth rates may be that in an exploited population the faster growing eels are more likely to reach maturity, so mature eels of the more exploited sex are likely to have grown faster than those of the less exploited sex.

Growth rate variation

The estimation method used permitted the variability in growth rate among eels to be modelled, by including a number of different growth rate cohorts. Because maturation is length-dependent, failing to model this variability would result in poor fit to the data, with too narrow distribution of maturation at age.

Local minima and optimisation

Using a multiple cohort approach led to a series of small peaks or local minima in the likelihood with respect to the growth rate parameters. This made it difficult to use gradient-based optimisation techniques, since they would become stuck at one of the local minima, and give inaccurate estimates of uncertainty using the Hessian matrix. This problem was avoided by using sampling-importance-resampling, a technique that estimates the likelihood at a range of proposal values, and is not affected by local minima as long as the proposal distribution covers an appropriate range of values. The most appropriate proposal distribution is the posterior. Appropriate proposal distributions were selected by trial and error and by developing a series of diagnostic graphing techniques to ensure that the proposal distribution was sampling an appropriate range.

One of the problems with SIR, however, is that it becomes highly computer-intensive as the number of parameters increases. The large number of parameters in this analysis made it necessary to do the analysis in stages, optimising over a few parameters at a time, and using the estimated posterior values as proposals for the combined stage. The computer-time problem was also negotiated by using the Condor distributed computing system (Tannenbaum, 2001), permitting the use of large numbers of computer CPUs at once. SIR is inherently suitable for parallel processing, since it

involves repeating the same analysis for hundreds of thousands of combinations of values. We used an array of up to 20 computers running Microsoft Windows 2000.

Differentiation

Differentiation was modelled as a function of length using the formula proposed by De Leo and Gatto (1995) for eel maturation. This permitted all three observed ‘sexes’ (male, female, and undifferentiated) to be included in the model, by modelling males and females and pooling their numbers when in the undifferentiated state. Length at differentiation is partly determined by how differentiation is defined, and also contains observation error as a function of the analyst’s technique and skill. For example, tissue histology would be able to determine the sex of differentiation at an earlier stage than examination of the gonad with the naked eye.

No statistically significant difference between the sexes in length at 50 per cent probability of differentiation was apparent for either Queensland or New South Wales (Tables 3 and 4) surprisingly, since Walsh *et al.* (2003) found that females differentiated at a larger size than males (mean 590 mm versus 520 mm) using the same New South Wales data. This difference probably results from the analytical techniques used. Walsh *et al.* (2003) used the observed lengths of individuals in stage 1 differentiation. We did not model stage of differentiation, but did account for losses of larger males due to emigration. This loss is likely to have been substantial enough to account for the difference between the sexes, since Walsh found maturation (440–620 mm) occurring very shortly after differentiation (420–600 mm). This demonstrates the potential for a mixture of processes to bias an estimated parameter, when the model used is too simple to describe the processes.

Table 3: Posterior parameter estimates for male, female, and undifferentiated eels caught during fishery-independent electrofishing sampling in southeast Queensland.

Parameter	Distribution	Estimated values		Posterior percentiles		
		p1	p2	2.5	50	97.5
a1	Normal	52.00	0.15	51.71	52.00	52.29
a2	Normal	0.44	0.01	0.41	0.44	0.46
b1	Normal	36.94	1.49	34.02	36.94	39.87
mat γ fem	Beta	20.45	80.71	0.13	0.20	0.29
mat λ fem	Normal	759.74	16.89	726.64	759.74	792.84
mat η fem	Gamma	0.98	12.02	0.28	8.06	43.73
mat γ male	Beta	11.80	1.21	0.71	0.93	1.00
mat λ male	Normal	532.72	8.41	516.23	532.72	549.20
mat η fem	Gamma	5.95	2.11	4.58	11.85	24.45
d γ	Beta	54.72	83.06	0.32	0.40	0.48
d λ	Normal	311.83	11.54	289.22	311.83	334.44
d η	Gamma	24.16	0.96	14.86	22.84	33.26
M	Normal	-1.59	0.11	-1.79	-1.59	-1.38
sex ratio	Normal	3.16	0.60	1.99	3.16	4.34
sel μ	Normal	172.82	4.40	164.19	172.82	181.45
sel σ	Normal	11.44	1.97	7.59	11.44	15.29

Table 4: Posterior parameter estimates for male, female, and undifferentiated eels caught during fishery-independent electrofishing sampling in New South Wales.

Parameter	Distribution	Estimated values		Posterior percentiles		
		p1	p2	2.5	50	97.5
a1	Normal	52.01	0.15	51.71	52.01	52.31
a2	Normal	0.34	0.01	0.32	0.34	0.35
b1	Normal	40.77	0.59	39.61	40.77	41.93
mat γ fem	Beta	0.50	0.00	0.50	0.50	0.50
mat λ fem	Normal	863.24	0.00	863.24	863.24	863.24
mat η fem	Gamma	126.86	0.00	126.86	126.86	126.86
mat γ male	Beta	0.97	0.00	0.97	0.97	0.97
mat λ male	Normal	527.54	0.00	527.54	527.54	527.54
mat η fem	Gamma	39.82	0.00	39.82	39.82	39.82
d γ	Beta	41.93	1.21	0.91	0.98	1.00
d λ	Normal	486.96	4.96	477.24	486.96	496.67
d η	Gamma	100.77	0.51	41.57	50.89	61.49
M	Normal	-6.26	1.08	-8.39	-6.26	-4.13
sex ratio	Normal	2.80	0.36	2.10	2.80	3.51
sel μ	Normal	562.41	6.01	550.63	562.41	574.20
sel σ	Normal	37.05	2.02	33.09	37.05	41.00

Table 5: Parameter estimates for Queensland age at length general linear model. Parameters for sites are differences compared to the reference site 1.

Parameter	Site name	Estimate	SE	t(90)	t pr
α		0.29	2.35	0.13	0.901
β		0.02155	0.00449	4.79	<.001
Site 1	Nindooindah	0			
Site 2	Darlington Park Bridge	2.28	1.54	1.48	0.143
Site 3	Chardons Bridge	-0.31	3.45	-0.09	0.928
Site 4	Sandy Creek	-5.56	3.45	-1.61	0.111
Site 5	Cedar Creek	6.28	1.97	3.18	0.002
Site 6	Canungra Creek	-0.96	3.46	-0.28	0.782
Site 7	Site 4	-1.12	1.73	-0.65	0.519
Site 8	Site 3	1.72	2.13	0.81	0.422
Site 9	Site 2	2.13	3.56	0.6	0.551
Site 10	Site 1	-2.53	3.46	-0.73	0.466
Site 11	Guanaba Creek Road	-3.17	2.25	-1.41	0.161
Site 12	Clagiraba Road	-0.24	2.11	-0.11	0.909
Site 13	Beechmont Road	-1.88	1.48	-1.27	0.207
Site 15	Prices Creek	-3.61	3.47	-1.04	0.302
Site 16	Guanaba Creek	-5.08	2.77	-1.84	0.069
Site 17	Site 2	-5.54	3.54	-1.56	0.121
Site 18	Site 1	-5.01	3.52	-1.42	0.159
Site 19	Site 5	-4.38	1.53	-2.86	0.005
Site 20	Site 4	-6.64	2.14	-3.1	0.003
Site 21	Williams Bridge	-1.35	2.62	-0.52	0.607
Site 22	Cusack Lane	-7.3	3.58	-2.04	0.045
Site 23	Sandy Creek	-8.1	3.45	-2.35	0.021
Site 25	Moy Pocket	-2.64	2.36	-1.12	0.267
Site 26	Kenilworth	-4.34	2.63	-1.65	0.103
Site 27	Morrisons Crossing	0.47	1.99	0.24	0.814
Site 28	Drapers Crossing	-5.51	2.7	-2.04	0.044
Site 29	Samford Creek	-7.25	2.66	-2.72	0.008
Site 30	Albany Creek	-4.47	3.5	-1.28	0.205
Site 31	Wilson's Pocket Road	6.99	3.52	1.99	0.05

Table 6: Parameter estimates for NSW age at length general linear model. Parameters for sites are differences compared to the reference site CC1.

Parameter	Site name		Estimate	SE	t(399)	t pr
α			-4.74	2.56	-1.86	0.064
β			0.03218	0.00349	9.23	<.001
Site No	CC	Clarence River 1	0			
Site No	CC	Clarence River 2	2.68	1.73	1.55	0.122
Site No	CC	Clarence River 3	-4.2	1.48	-2.84	0.005
Site No	CC	Clarence River 4	2.79	1.75	1.6	0.111
Site No	CC	Clarence River 5	-1.77	2.33	-0.76	0.448
Site No	CC	Clarence River 6	-4.14	1.34	-3.1	0.002
Site No	CC	Clarence River 7	-2.47	4.05	-0.61	0.543
Site No	CC	Clarence, Orara River	2.9	1.86	1.56	0.119
Site No	HH	Hacking River	5.4	4.05	1.33	0.183
Site No	HH	Kangaroo and Hacking	3.21	1.42	2.27	0.024
Site No	HH	Port Hacking	0.97	1.47	0.66	0.512
Site No	NH	Berowra Ck	-0.3	3.01	-0.1	0.92
Site No	NH	Hawkesbury River 1	0.01	1.37	0.01	0.994
Site No	NH	Hawkesbury River 2	1.57	1.49	1.05	0.295
Site No	NH	Hawkesbury River 3	4.06	1.82	2.23	0.026
Site No	NH	Hawkesbury River 4	0.36	1.82	0.2	0.844
Site No	NH	Murrumurra Ck	-0.66	1.94	-0.34	0.734
Site No	NH	Nepean River 1	7.31	1.44	5.07	<.001
Site No	NH	Nepean River 2	-0.17	2.56	-0.06	0.949
Site No	RS	Clarence River – rivers survey	2.48	1.95	1.27	0.204
Site No	RS	Gara River	-0.9	3.01	-0.3	0.765
Site No	RS	Macleay River	-0.35	2.59	-0.13	0.894
Site No	RS	Myall River	-5.65	4.06	-1.39	0.164
Site No	WO	Wallamba River	-1.21	2.58	-0.47	0.638

Analysis of the Queensland data, for which full selectivity occurs well below the length at differentiation (Table 3), also found no difference between the sexes. Histological studies have found that the environment can affect both length at differentiation and sex ratio, with ovaries found in shorter eels in stock with a prevalence of females (European eel) (Colombo, 1996). This suggests that females might differentiate at a smaller size than males, but we found no evidence for this in *A. reinhardtii*. Previous studies have found both males and females differentiating over a wide size range (Walsh *et al.*, 2003, Table 5), with females sometimes differentiating larger than males. Such differences may be observed due to a combination of sampling in habitats with different sex ratios, selectivity, and the effects of emigration.

Sex ratio

Sex ratio at differentiation can be inferred directly from the estimated sex ratio parameter. The model structure requires sex ratio to be set at age zero, but sex ratio does not change between age zero and differentiation.

Conclusions

Investigating complex life cycles like those of anguillid eels requires a clear understanding that interactions among the processes can affect observations. For example, length-related emigration due to maturation can affect growth rate estimates; selectivity can affect estimates of length at differentiation; and fishing pressure can affect many parameter estimates including growth rate and length at maturity. The model used in this analysis integrated many of these processes, and provided unbiased estimates of a number of population parameters and their variability.

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Chapter 6: Modelling the effects of commercial harvest on Australian longfinned eel (*Anguilla reinhardtii*) stocks

Simon Hoyle

Abstract

Freshwater eels in Australia are exploited by commercial fisheries at the adult and glass eel stages. A model was developed to investigate the effect of the adult commercial eel fishery on escapement of migrating longfin (*Anguilla reinhardtii*) eels from New South Wales and southern Queensland. As a result of market requirements and sexual dimorphism, the yellow eel fishery for this species largely captures females. Exploitation of yellow eels, without closed areas, at rates of 5 per cent and 10 per cent per year reduced the egg production per recruit of longfin females by 57 per cent and 80 per cent respectively below unexploited levels. The model suggests that, within a population subunit, management of Australian longfin eels for egg production conflicts with management for fishing yield. Access of glass eels during upstream migration is also likely to reduce egg production across large areas. Separation into areas for egg production and areas for fishing yield, combined with restocking or improvements in eel access into catchments with significant barriers to migration, is likely to achieve better outcomes for both. Male eels are shorter and younger at spawning and therefore spend less time vulnerable to fishing. Males are also more likely to occur lower in the catchment where access is less affected by dams and weirs. However, males are generally not targeted by the export fishery for yellow eels because most males are below the minimum market size of 500 g.

Project objectives

Develop a management population dynamics model and use it to investigate management options. Assess the applicability of the above techniques to other eel fisheries in Australia, in collaboration with NSW. Distribute developed tools via the Australia and New Zealand Eel Reference Group.

Introduction

An eels' complex life history involves a marine larval phase (the leptocephalus), arrival in estuaries and metamorphosis (the glass eel), development of pigmentation and growth in fresh water for between about five and 100 years (the elver through to the adult/yellow eel), followed by sexual maturation (the silver eel) and migration to the sea to breed. In the various international fisheries, exploitation can take place at any or all of the post-larval stages. In Australia exploitation occurs at glass eel and yellow eel stages.

In eel species with extensive recruitment data (*Anguilla japonica*, *A. anguilla*, *A. rostrata*), the numbers of migrating glass eels (juveniles recruiting to fresh water from the sea) have declined severely in recent years (Gascuel, 1987; Moriarty, 1990; Dutil *et al.*, 1989; Castonguay *et al.*, 1994). Suggested causes of the declines include commercial fishing of elvers and glass eels, overfishing of adult stocks, chemical contamination, habitat modification (such as construction of in-stream barriers), oceanic changes and irruptions of disease or parasites (Castonguay *et al.*, 1994; Dekker 2000; Dekker, 2003).

Egg production per recruit, or spawning per recruit, is vulnerable to exploitation of the adult stock, since eels breed late, and only once. Unlike most other species, eels contribute to either spawning or

fishing yield; it is not possible to follow the management practise in many other Australian fisheries and permit eels to breed once before capture.

Eels mature late and breed only once, making them vulnerable to overexploitation. Female age at spawning varies between species and locations, with ages observed between 5 and 39 years for *A. reinhardtii* in this study in Queensland, 10 to 52 years for *A. reinhardtii* in New South Wales (Walsh *et al.*, 2004), and with mean ages from 7.9 years for *A. anguilla* in the Imsa river, Norway (Vollestad and Jonsson, 1988), to 93 years for *A. dieffenbachii* in the subalpine Lake Rotoiti, New Zealand (Jellyman, 1995). Late-maturing species are especially vulnerable to overfishing (Sadovy, 2001). For example, maturity at over 18 years is a common trait of species at significant risk of extinction, including *A. rostrata*, in the Great Lakes–St Lawrence biozone (Parent and Schriml, 1995). Concern has been expressed about the possible effect of fishing on the number of spawning females for a number of anguillid species, including *A. dieffenbachii* (Chisnall and Hicks, 1993; Jellyman, 1995; Hoyle and Jellyman, 2002), *A. rostrata* (Casselman 2003; Casselman and Cairns, 2003), and *A. anguilla* (Dekker, 2000 and 2003). Changes in numbers of migrating eels may be difficult to observe due to insufficient monitoring and great natural variability.

There is some evidence of an association between reduced spawning and reduced recruitment (Dekker, 2003), although alternative hypotheses have been advanced to explain recruitment declines (Castonguay, *et al.* 1994; Kirk 2003; Knights 2003). Nonetheless, the dramatic declines of many eel species worldwide (e.g. *A. anguilla* now outside safe margins (ICES, 2004)) are sufficient to cause great concern (Casselman and Cairns, 2003), and to highlight the advisability of taking a precautionary approach to eel management (Russell and Potter, 2003).

Exploitation normally reduces egg production, and moderate reductions are not usually a problem, as long as egg production remains above the replacement level. Exploitation beyond this level is one definition of recruitment overfishing (Sissenwine and Shepherd, 1988). The appropriate level is difficult to determine. It varies between taxa and populations and through time and has not been determined for any eel species. Studies of a range of European and North American fish stocks have identified a conservative level as 30 per cent of unfished spawning per recruit — at which 80 per cent of studied stocks were able to replace themselves (Mace and Sissenwine, 1993) — or 50 per cent of r_{max} (Myers *et al.*, 1994). However, the late age at maturity of eels makes the appropriateness of these estimates uncertain. Late-maturing long-lived animals have low intrinsic rates of increase and therefore low resistance to ‘extraordinary’ mortality (Musick, 1999a,b).

There is some uncertainty about the degree to which panmixia occurs in eel populations; this also applies to Australian longfin eels. Most eel species have long been assumed to constitute single interbreeding populations (panmixia), geographically subdivided in the growing freshwater phase but coming together to breed at (largely unknown) sites in the deep ocean. However, recent genetic work has found genetic subdivision among regions for European eels (Anon. 2000; Daemen *et al.*, 2001; Wirth and Bernatchez, 2001), and there is a suggestion of similar subdivision within *A. reinhardtii* (Andrew Moore, pers. comm.).

Subdivision of this kind would require managers to consider escapement on the scale of the subdivision. Subdivision at the catchment level would require each catchment to be managed separately, since each catchment would rely on its own egg production for future recruitment. However, the subdivision reported in Europe is on a large scale, with no suggestion of subdivision at a catchment level. The oceanographic mechanisms supporting recruitment within Australia — advection by the east Australian current — make subdivision unlikely on any but the broadest scale. The most plausible subdivision is between New Caledonian and Australian populations, and/or between northern Queensland and the rest of eastern Australia.

In this chapter, I use a simulation model to estimate the effect of fishing on the spawning per recruit of southern Queensland and New South Wales populations of female and male *A. reinhardtii*, and on fishing yield per recruit. I compare the impacts of various management measures. The management regimes I investigate relate to those already in practice in Queensland and New South Wales, being the value of minimum legal size (currently 30 cm), and the proportion of productive eel habitat to which eels have access.

I also examine the effect of various exploitation rates, including zero exploitation in closed areas. The panmictic nature of eel populations gives closures particular power, since recruitment sourced from the reserve is distributed throughout the species' range.

The long-term (equilibrium) effect of area closures is estimated, very simply, by considering the effect of returning a proportion of the fishery's production to an unfished state. Since all estimates in this study are made relative to this unfished state, modelling is unnecessary.

Previous population models of anguillid fisheries have focused on maximising fishing yield using either a yield-per-recruit approach (Gascuel and Fontenelle, 1994), or a more sophisticated demographic modelling approach (De Leo and Gatto, 1995 and 1996). Dekker (2000) has addressed the effect of fishing on spawning of *A. anguilla* in a heavily exploited fishery, Lake IJsselmeer in the Netherlands. Hoyle and Jellyman (2002) modelled the impact of commercial harvest on adult eel escapement and fishing yield for New Zealand longfin and shortfin eels. They adapted Francis and Jellyman's (1999) model that was designed to look at the ability of length monitoring to detect changes in the exploitation rate of the commercial fishery.

The modelling exercise presented here is closely based on this published application for New Zealand eels (Hoyle and Jellyman, 2002).

The model used in this chapter has been developed into a user-friendly application for managers to work with, and can be found on the associated compact disk, or downloaded from www.dpi.qld.gov.au/fishweb/. Use of this model is described by Hoyle in Chapter 7 of this report.

Methods

The model is essentially the model used in Hoyle and Jellyman (2002), with a few minor changes. Much of the text of that paper's methods section is reproduced below, with appropriate modifications. The model takes a per-recruit approach. It incorporates variation among individual eels in growth and initial length by modelling twenty growth rate classes, with growth rates ranging from slow to fast across a normal distribution. It models maturity using the approach of de Leo and Gatto (1995). It does not model stochasticity in recruitment, or model more than one cohort, since I am concerned with long-term effects of management strategies, rather than uncertainty or short-term responses.

The data come from Chapter 5 of this report, and are summarised in Table 1.

Table 1: Model parameters for both sexes, and for both states unless stated otherwise (after Francis and Jellyman, 1999; and Todd 1981).

Type	Parameter	Qld F	Qld M	NSW F	NSW M
Growth	a1	5.2	5.2	5.2	5.2
	a2	0.44	0.44	0.34	0.34
	b1	3.69	3.69	4.08	4.08
Length-weight	α	2.72×10^{-3}	2.72×10^{-3}	2.72×10^{-3}	2.72×10^{-3}
	β	3.016	3.016	3.016	3.016
Length fecundity	c	2.66×10^{-2}		2.66×10^{-2}	
	d	3.59		3.59	
Natural mortality	M (year ⁻¹)	0.04	0.04	0.04	0.04
Maturity	γ (no units)	0.2	0.93	0.5	0.97
	λ (cm)	75.974	53.272	86.32	52.754
	η (cm)	0.806	1.185	12.686	3.982

Basic assumptions

The probability that an eel of length L will become mature (μ_L) is calculated by the equation:

$$\mu_L = \frac{\gamma}{1 + \exp\left(\frac{\lambda - L}{\eta}\right)} \quad (\text{Equation 1}) \quad (\text{de Leo and Gatto, 1995})$$

where γ is the maximum rate of metamorphosis, λ is a semi-saturation constant, and η is inversely proportional to the slope of the metamorphosis curve at $L = \lambda$. Parameters for the model are those estimated in Chapter 5.

Additional parameters were estimated for fecundity at length, based on silver eel data collected during the eel project. Gonads of female eels were extracted and weighed to the nearest gram. The length-weight relationship was estimated using least squares as $W = cL^{d+\epsilon}$.

Estimates of M have not been obtained for *A. reinhardtii*. We used two alternative estimates of 0.04 and 0.08 years⁻¹ for all age classes, both sexes, and both states. Various exploitation rates were explored; this parameter is quite uncertain and clearly varies considerably between areas on a variety of spatial scales.

The length L of an eel of age i is calculated by

$$L = (a_1 + b_1 i) e^{a_2 \cdot \epsilon} \quad (\text{Equation 2}),$$

where the parameters a_1 and b_1 define mean length as a linear function of age, a_2 describes the standard deviation of length-at-age, and ε is a standard normal variate representing between-individual variation in length at age (Francis and Jellyman 1999). The twenty values of ε used to define the growth classes were the $j/40$ quantiles of the standard normal distribution, where $j = 1, 3, 5, \dots, 39$. Linear growth appears to be characteristic of eel species (Jellyman, 1997; Graynoth and Taylor, 2004). Jellyman (1995) has suggested that eels' lack of investment in annual spawning leads to linear growth.

Weight is calculated by $W = \alpha L^\beta$.

Exploitation rate at length L E_L is calculated using (Equation 3):

$$E_L = a + b(L - L_{\text{MinLW}}) \quad \text{if minimum legal size} < L < \text{maximum legal size}$$

$$E_L = 0 \quad \text{otherwise,}$$

where a and b are the intercept and slope of the size selectivity function. The size selectivity of the fishery overall has not been estimated, so a flat selectivity curve was assumed. However, the sensitivity of this assumption was investigated (see below).

The number of eggs in metamorphosed eels is calculated using the equation $G_L = cL^d$, where c and d are the parameters of the length-fecundity equation for the species.

The cohort of eels is followed from the age of one, when both maturity and exploitation are minimal, to age 100, when insignificant numbers remain.

Demographic model

Given the number $N_{i,c}$ of yellow pre-reproductive eels at age i in growth class c , I compute, in the following order:

1) eel length $L_{i,c}$ by using eq. 2, eel weight $W_{i,c}$ at length $L_{i,c}$, and number of eggs $G_{i,c}$ at length $L_{i,c}$ assuming metamorphosis.

2) number of eels undergoing metamorphosis at age i in growth class c

$$M_{i,c} = \mu_L N_{i,c}$$

number of eels remaining after metamorphosis

$$N_{i,c}^* = (1 - \mu_L) N_{i,c}$$

3) number of eels surviving natural mortality

$$N_{i,c}^+ = (1 - e^{-M}) N_{i,c}^*$$

4) number of eels harvested at age i in growth class c , given E_L , the exploitation rate at length:

$$Y_{i,c} = E_L N_{i,c}^+$$

number of eels escaping fishing and making up the following year's yellow stock:

$$N_{i+1,c} = (1 - E_L) N_{i,c}^+$$

Thus $N_{i+1,c} = N_{i,c} \cdot (1 - \mu_L) \cdot e^{-M} \cdot (1 - E_L)$ ($i = 1, 2, \dots, i_{\text{max}}, c = 1, 2, \dots, c_{\text{max}}$)

The overall yield is thus given by the contribution by all age classes and growth classes of legal size, namely

$$Y^\# = \sum Y_{i,c} \quad (i = 1, 2, \dots, i_{\text{max}}, c = 1, 2, \dots, c_{\text{max}})$$

while overall biomass is:

$$W^{\#} = \sum W_{i,c} Y_{i,c} \quad (i = 1, 2, \dots, i_{max}, c = 1, 2, \dots, c_{max})$$

Metamorphosing biomass is:

$$MB^{\#} = \sum W_{i,c} M_{i,c} \quad (i = 1, 2, \dots, i_{max}, c = 1, 2, \dots, c_{max})$$

Eggs in metamorphosing eels are:

$$G^{\#} = \sum G_{i,c} M_{i,c} \quad (i = 1, 2, \dots, i_{max}, c = 1, 2, \dots, c_{max})$$

Decision variables

Any management policy simulated by the model is defined by a combination of 1) fishing mortality, 2) minimum legal size, and 3) maximum legal size. The model is used to investigate the effect of management alternatives for females and males in Queensland and New South Wales on relative spawning per recruit and relative yield per recruit. The word ‘relative’ is used because performance can only be assessed against the unexploited population. Total biomass, total egg production, and even total equilibrium yield are unknown, since the natural population is not at equilibrium.

I examined the effect of the following management measures: minimum legal size of 30 cm, and possible higher values; and various rates of exploitation. For each component of the eel fishery, I estimated the minimum legal size that would maximise yield-per-recruit at a range of exploitation rates, and the exploitation rate that would maximise yield-per-recruit at the current minimum legal size (30 cm). I compared all yields to the 30 cm legal size limit, although in practice for many fishers the size limit is effectively 58 cm because of the commercial size required. Some fishers in New South Wales catch eels below 58 cm for aquaculture grow-out.

I also investigated the sensitivity of the results for females to two alternative selectivity functions, since it is possible that catchability of eels in eel traps increases with size. In the standard (no size selectivity) model, the selectivity intercept variable a is set to 0.05, with $b = 0$. For the alternative selectivity I assumed that catchability was proportional to length, on the assumption that larger eels are behaviourally dominant and more likely to encounter fishing gear. I investigated the way this assumption affected spawning per recruit. In this model, a is set to 0.02, and b is adjusted so that the average exploitation rate for ages 26 to 45 is the same as observed when $a = 0.05$ and $b = 0$.

Results

Females — Queensland and NSW

At the current minimum legal size of 30 cm, the predicted egg production per recruit for Queensland was reduced by 57 per cent and 80 per cent from unfished levels at annual exploitation rates of 5 per cent and 10 per cent respectively (Figure 1a). The equivalent figures for New South Wales were 46 per cent and 70 per cent. There was less reduction in egg production per recruit with a 50 cm minimum legal size, which reflects more closely the actual selectivity in the New South Wales fishery; reductions in Queensland at 5 per cent and 10 per cent exploitation were 56 per cent and 70 per cent, and in New South Wales were 33 per cent and 54 per cent respectively (Figure 2a).

The optimum yield-per-recruit at the current minimum legal size was obtained with a moderate exploitation rate of between 10 per cent and 15 per cent. However, yield at these exploitation levels was increased by raising the minimum legal size, which reflects the actual selectivity of the New South Wales fishery. A large proportion of this selectivity is achieved by voluntary release of eels smaller than 500g (58 cm mean length) not the mesh size of the traps (Pease pers. comm.). The optimum minimum legal size increased as the exploitation rate increased (Figures 1b and 2b).

optimum minimum legal size increased as the exploitation rate increased (Figures 1b and 2b). In New South Wales at an exploitation rate of 5 per cent, the optimum minimum legal size was 40 cm, which raised yield-per-recruit by only 3.6 per cent. However, at an exploitation rate of 10 per cent the optimum minimum legal size was 50 cm, increasing yield-per-recruit by 58 per cent. At exploitation rate of 20 per cent, the optimum legal size of 55 cm raised yield by 113 per cent over the yield at 30 cm. Capture of female eels at smaller sizes for aquaculture should not necessarily be interpreted as reducing yield however. On balance they may supply as much or more economic benefit in terms of fishing yield and employment as those that remain in the wild.

Males — Queensland and NSW

At the current minimum legal size of 30 cm, the predicted spawning biomass per recruit for Queensland was reduced by 27 per cent and 46 per cent from unfished levels at annual exploitation rates of 5 per cent and 10 per cent respectively. The equivalent figures for New South Wales were 27 per cent and 40 per cent (Figures 3a and 4a). There was less reduction in spawning biomass per recruit with a 50 cm minimum legal size, which reflects more closely the actual selectivity in the New South Wales fishery; reductions in Queensland at 5 per cent and 10 per cent exploitation were 4.5 per cent and 9 per cent, and in New South Wales were 3 per cent and 6 per cent respectively.

Even at 50 per cent fishing pressure New South Wales spawning biomass from fished areas was only reduced by 26 per cent. It is also likely that a large proportion of the male population is in unfished areas, since males tend to dominate lower down in catchments (unfished in Queensland and less subject to migration barriers), and in Queensland. Sex ratio in Queensland was estimated as 3.35 males to 1 female (Chapter 5). Fishing pressure is unlikely to be significantly affecting male spawning biomass.

The yield of male longfins was not increased by raising the minimum legal size, unless exploitation rate was 15 per cent or more (Figures 3b and 4b), and then only for small increases to 35 cm. At an exploitation rate of 5 per cent, a minimum legal size of 50 cm (reflecting actual selectivity in New South Wales) lowered yield-per-recruit by 73 per cent compared to the current legal size; and at an exploitation rate of 10 per cent a minimum legal size of 50 cm gave 67 per cent less yield-per-recruit than at 30 cm. An increased size limit decreases the harvest of the dimorphically smaller males.

At the current minimum legal size, the maximum yield was obtained at a high 35 per cent annual exploitation rate (Figures 3c and 4c). Maximum legal size had no significant effect on yield of male longfins. Thus the male and female components of the longfin fishery respond very differently to exploitation.

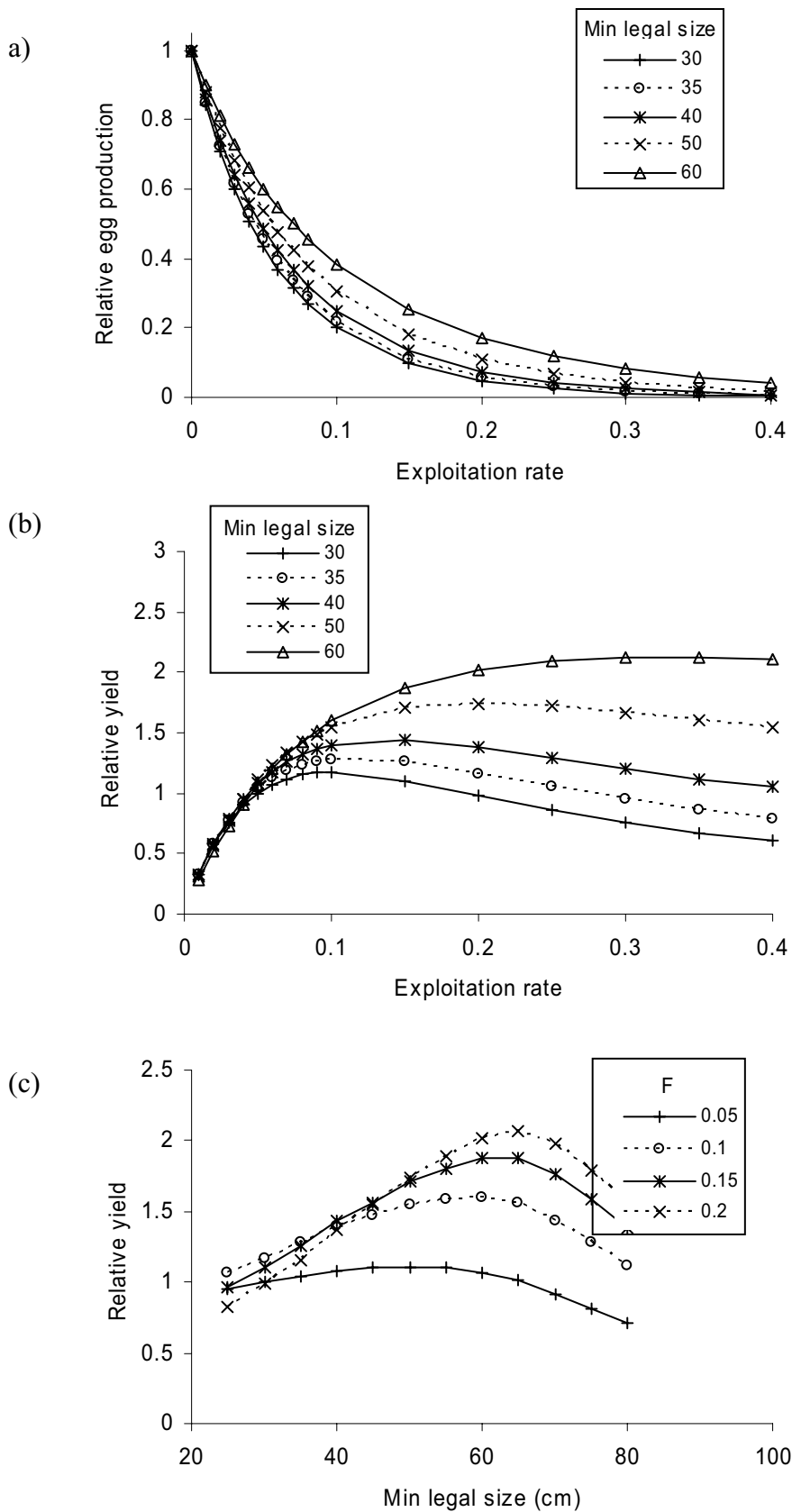


Figure 1: Queensland female longfin eels at equilibrium: Relative egg production per recruit and yield per recruit at a range of exploitation rates and minimum legal sizes. a) Relative egg production per recruit. b) Relative yield-per-recruit declines at higher exploitation rates, depending on legal size. Minimum legal size affects yield-per-recruit more at high fishing pressures. c) Relative yield-per-recruit is greatest at high legal sizes and fishing pressures.

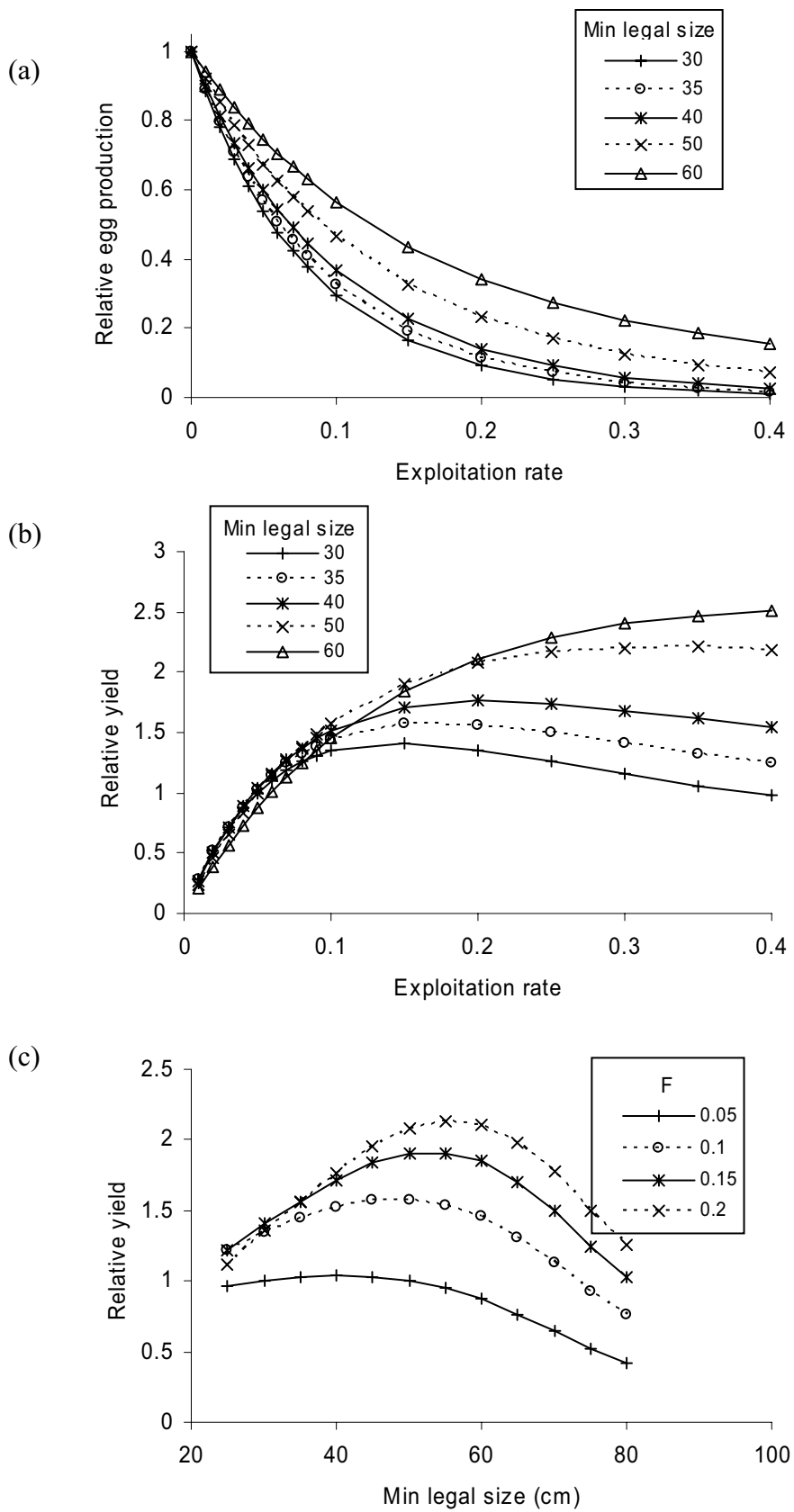


Figure 2: NSW female longfin eels at equilibrium: Relative egg production per recruit and yield per recruit at a range of exploitation rates and minimum legal sizes. a) Relative egg production per recruit. b) Relative yield-per-recruit as a function of legal size and fishing pressure. c) Relative yield-per-recruit at a range of minimum legal sizes and exploitation rates.

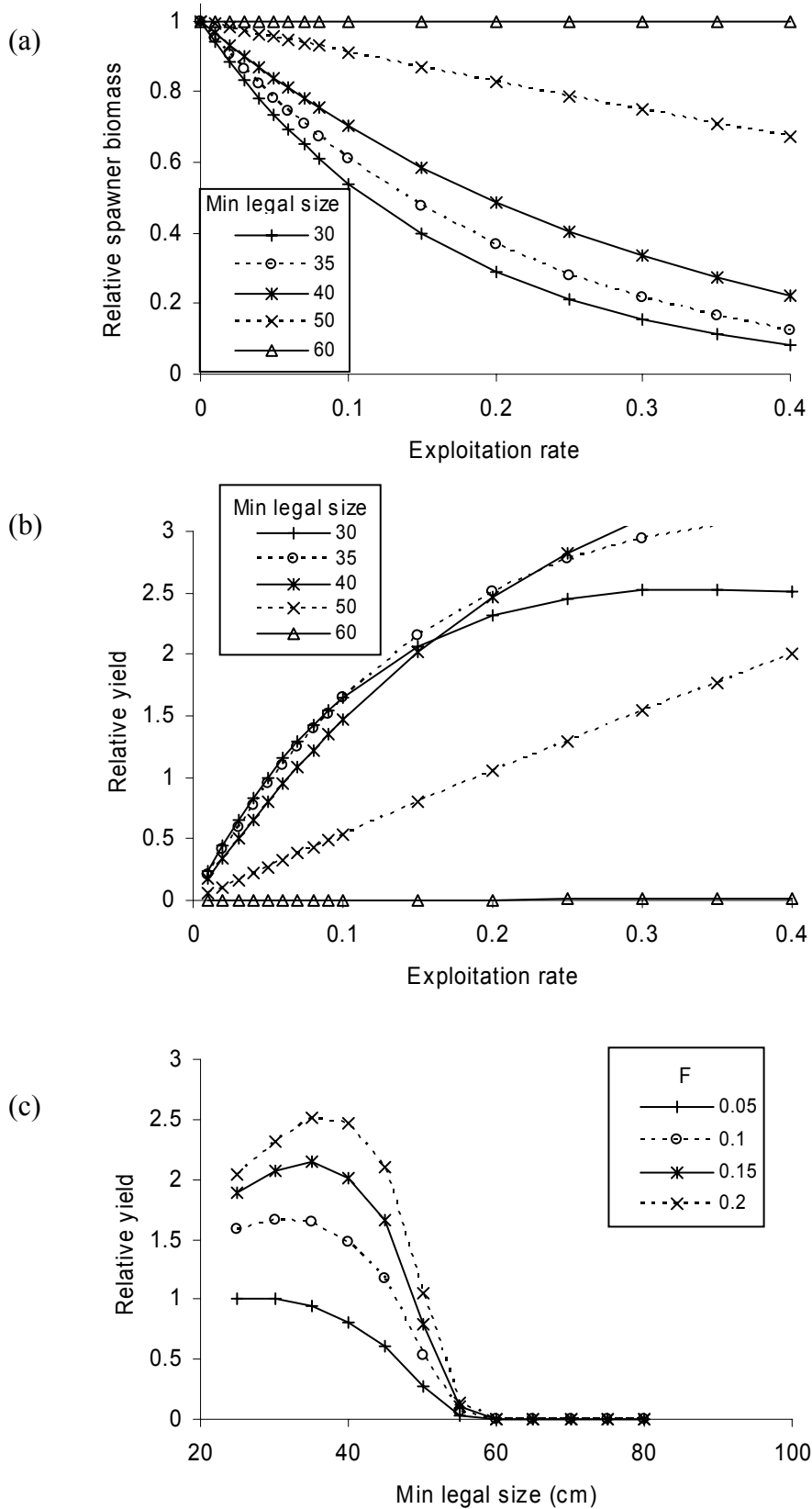


Figure 3: Queensland male longfin eels at equilibrium: Relative spawning biomass per recruit and yield per recruit at a range of exploitation rates and minimum legal sizes. a) Relative spawning biomass per recruit. b) Relative yield-per-recruit increases with exploitation rate for all minimum legal sizes. c) Relative yield-per-recruit is greatest at high fishing pressure.

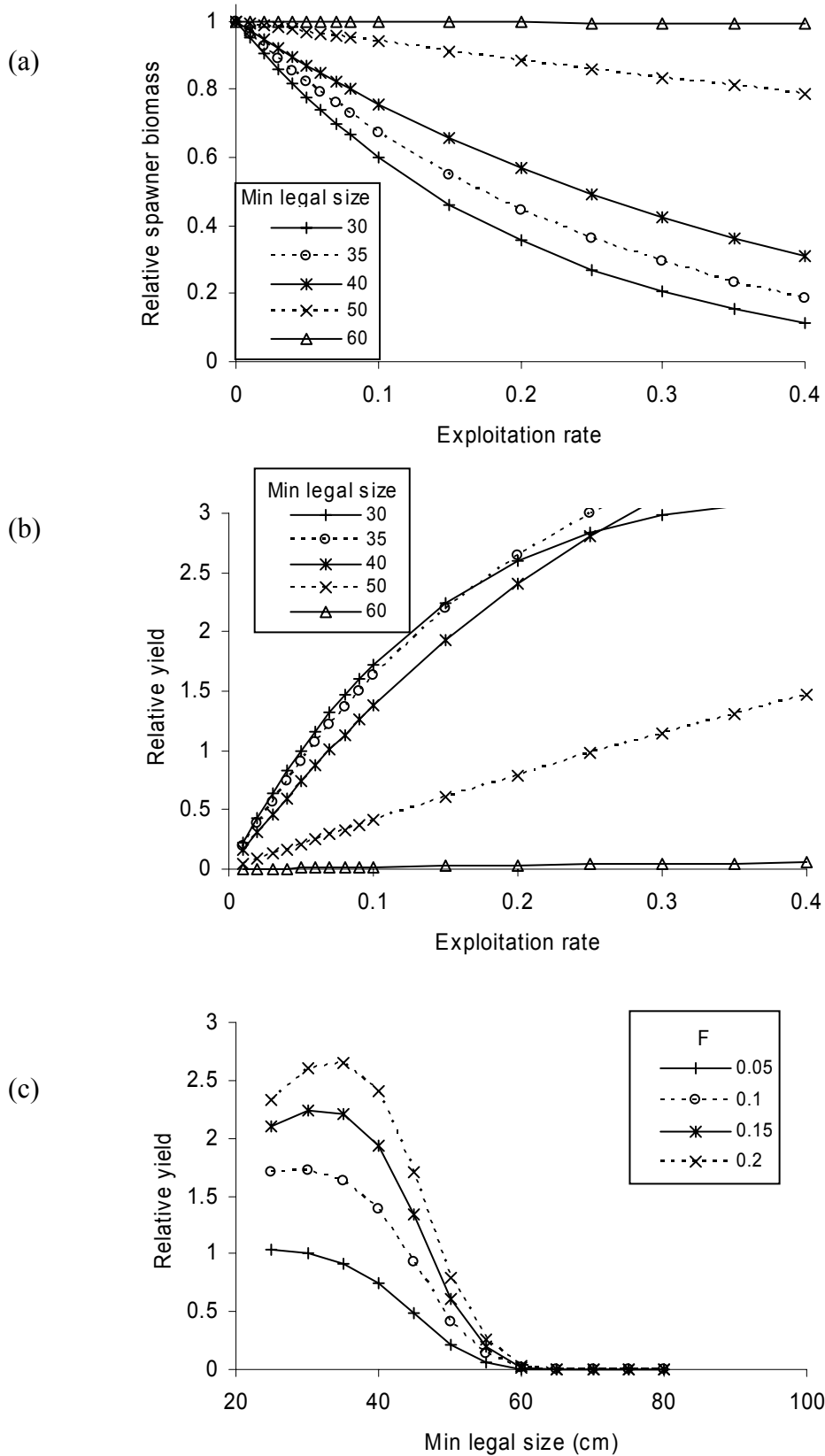


Figure 4: NSW male longfin eels at equilibrium: Relative spawning biomass per recruit and yield per recruit at a range of exploitation rates and minimum legal sizes. a) Relative spawning biomass per recruit. b) Relative yield-per-recruit increases with exploitation rate for all minimum legal sizes. c) Relative yield-per-recruit is greatest at high fishing pressure.

Discussion

The model used here predicts that moderate levels of fishing have the potential to reduce the egg production of fished waters. Results imply that female spawning biomass is likely to be reduced considerably more than male spawning biomass. Areas managed for optimal yield will have high fishing pressure and little egg production.

Estimates of spawning stock biomass have not been attempted for *A. reinhardtii*, but given the longevity of the species and the evidence of decline in other species of *Anguilla*, management should be conservative. Our models suggest that two main factors threaten eel egg production: yellow eel fisheries and in-stream barriers. Glass eel harvest could potentially also threaten recruitment, but evidence presented in Chapter 1 suggests that current levels of harvest are having no impact. Current harvest rates are probably not at a level where density dependent effects come into play. However exclusion of large numbers of glass eels by barriers and associated high mortalities at barrier sites means that areas upstream of major dams are probably well below their carrying capacity for eels (e.g. EHMP, 2004). The combined effect of barriers and yellow eel fisheries may prevent the majority of longfin females from spawning, despite the lack of a fishery targeting migrating eels. Given the high but unknown prevalence of in-stream barriers throughout eastern Australia, egg production from locations that eels can access should be maintained at high levels.

Managing to ensure adequate spawning stock

Given their life history, eels can be recruitment overfished without being growth overfished. It is unclear what proportion of the unfished egg production will maintain recruitment at replacement levels. This question has been addressed for a range of stocks with the conclusion that, where there is no basis for estimating the replacement level of spawning per recruit, 30 per cent of the unfished level is a conservative default value (Mace and Sissenwine, 1993).

Female eels are seen as more vulnerable than males since they live much longer before spawning; are found preferentially in upper catchment areas to which access is restricted by in-stream barriers; and experience far greater fishing pressure due to their larger size at migration. Estuarine areas are fished in New South Wales although not in Queensland, but most males migrate before becoming large enough to have commercial value.

Given the particular vulnerability of long-lived species, and the uncertainty about the current level of longfin depletion, a more conservative level such as 50 per cent of unfished spawning per recruit may be justified (suggested by Jellyman, 1993 with respect to escapement of migrating longfin females in New Zealand). Five mechanisms to attain such goals are considered in the light of the modelling results presented above. The mechanisms are: the imposition of legal sizes (minimum and maximum), control of exploitation rate, long-term spatial closures, restocking, and removal of barriers to recruitment and spawning emigration. The reader can manipulate these mechanisms in the attached eel management modelling CD. Instructions for its use are contained in Chapter 7.

Minimum legal size

The vulnerability of longfinned eels, especially in the years before they reach maturity, is of concern. A minimum legal size that would protect significant numbers of female spawners would exceed the length at migration of all males, which would therefore be lost to the fishery, although loss of males from the fishery may not be important as most males are below market size. It is not practical to have a different minimum legal size for the different sexes, for administrative and enforcement reasons, and because fishing equipment does not distinguish between the sexes. Management could use spatial closures to take advantage of the spatial separation often seen between the sexes, though again this would require there to be a market for small eels.

Raising minimum legal size would cause few difficulties for most fishers since the fishing gear currently used reaches full selectivity at about 55 cm (New South Wales result, Hoyle and Peel Chapter 5 this report). However, given this situation there also seems to be little need to change the regulations. Raising the legal size would prevent a small number of fishers from taking smaller eels for grow-out, with little benefit for sustainability.

Maximum legal size

Maximum legal size of 4 kg has been applied in New Zealand for longfin eels. The principle behind it is to allow 50 per cent escapement of migrating eels (Jellyman, 1993). Fecundity increases with length more steeply than does size (see Hoyle and Peel Chapter 5 this report), so larger eels are disproportionately valuable for egg production.

Maximum legal size has potential to protect longfin females effectively, but to do so it would have to be set quite low. Determining a suitable level would require more data on catchability by size. However, there are practical obstacles to such a management measure. Larger eels tend to be more valuable per unit weight than smaller eels in some markets, so this option is somewhat inefficient. If, as is probable, catchability increases with size, large longfin eels would be caught more frequently than smaller eels, with attendant likelihood of handling mortality. In addition, a narrow window of catchable lengths would give fishers an economic incentive to fish harder within the window, to catch eels before they grow beyond the size limit. This would reduce the efficiency of the fishery and still result in few eels reaching maturity.

Control of exploitation rate

A lowered exploitation rate could be used to increase the escapement of female longfin spawners, but a meaningful increase would require a substantial reduction throughout the fishery. Our models suggest that spawning per recruit of longfin females (from fished areas with full access to the sea) would reach 50 per cent of unfished levels at a long-term exploitation rate of approximately 6 per cent (Queensland) or 9 per cent (New South Wales), assuming full selectivity at 50 cm. A quota system could be used to manage the exploitation rate. However, quota systems are expensive to implement and difficult to manage across large subdivided fisheries. In addition, as I discuss below, those components of the fishery would actually yield better at higher exploitation rates.

Unfished areas

Closed areas are currently attracting much attention in fisheries management. Modelling suggests that marine reserves are most useful for species that remain within reserve boundaries, and have abundant larval spillover (Sladek Nowlis and Roberts, 1999; Sladek Nowlis, 2000). Reserves can help prevent recruitment overfishing, even when fishing mortality outside the reserve cannot be well controlled (Mangel, 2000), and total stock size is highly uncertain (Walters, 2000). Eel biology fits the former criteria well, with their panmixis, and yellow-phase territoriality and subdivision into catchments; and the latter criteria of uncertain stock size and fishing mortality rate are certainly true also.

At present exploitation rates, our models suggest that the fishery combined with in-stream barriers may be able to severely reduce egg production from fished waters. Thus, most egg production must come from eels from unfished areas with access to the sea. The most appropriate way of ensuring adequate escapement of migrants may be to establish and maintain unfished areas (Jellyman, 1993; 1995; Hoyle and Jellyman, 2002). Areas where growth rates are high, where longfins dominate, and where most eels are female, would be the most appropriate areas for closure. Females tend to be

more dominant in the upper catchment, where they are likely to experience access problems due to barriers.

Queensland already has closures on riverine areas and estuaries, with the fishery restricted to impounded waters. NSW has created refugia for female eels by closing all non-tidal flowing fresh waters to commercial fishing in the early 1900s to protect trout. It is assumed most females can migrate out through the commercially fished estuaries because outmigrating silver eels generally do not enter baited traps and the market is not for silver eels.

Restocking and removal of barriers

Restocking areas with glass eels or elvers or facilitating access to areas that have poor access for glass eels may increase egg production if mature females are able to escape. Large reservoirs are often difficult or impossible for glass eels and elvers to enter, and could be restocked or provided with elver passes. However, unless silver females can escape there will be no benefit for egg production. Smaller barriers that do not completely block glass eels may nevertheless significantly increase mortality. Removing these barriers or providing elver passes may increase the glass eel supply to all areas upstream. Such small barriers are less likely to pose significant problems for downstream migration, so attention to these barriers may be more beneficial for long-term egg production.

Managing to optimise yield

Setting aside large productive areas for egg production represents a significant opportunity cost to the fishery. However, if sufficient areas are set aside to guarantee egg production, the remaining areas can be managed to maximise fishing yield. Because species composition, sex ratio, and growth rate vary among areas, management must also vary among areas if it is to optimise yield.

Minimum legal size

Improvement of yield in the long term could theoretically be achieved by increasing the size at which female longfinned eels are taken. This would be impractical to achieve with regulations, because the ideal minimum size to maximise yield depends on the sex ratio and species composition of the area, as discussed above, and the fishing gear does not distinguish between sexes. However, if male eels were not considered valuable to the fishery then a minimum size limit that excluded most males could be one option. The current commercial size requirements appear to be quite effective for generating high yields if fishing pressures are high enough.

Exploitation rate

At the current minimum legal size, the best yield from female longfins would be obtained with a comparatively low exploitation rate, between 5 per cent and 8 per cent. However, at the current commercial size, the best yields are taken above 15–20 per cent exploitation rate. Lower exploitation rates allow female longfins to reach larger sizes, at which size weight gain is most rapid given a linear length growth trajectory. On the other hand, the yields of male longfins tend to increase under greater fishing pressure. Therefore the appropriate exploitation rate depends on the sex ratio of a particular population within the fishery.

Restocking and removal of barriers

Restocking areas with poor access for glass eels, or facilitating access, will increase the productivity of these areas for fishing. Large reservoirs are often difficult or impossible for glass eels and elvers to enter, and could be restocked or provided with elver passes. Since their barriers to downstream migration make them less likely to contribute to egg production, they could be ‘farmed’ with high fishing pressure and high size at full selectivity, to maximize yield. Some eels placed in these

reservoirs may migrate further upstream to unfished areas and may be able to contribute to egg production on those occasions when the dam spills. Some reservoirs are also closed to commercial fishing by the water management authorities. Providing eel passage into these areas would increase the standing stock of potential spawners, and these will be able to escape to spawn during overflow events.

Glass eels and elvers for restocking should be collected immediately below the wall of the reservoir into which they are to be released. This will ensure that it is only eels trying to enter the upstream area that are able to do so. Without collection and transport of these eels, it is highly likely that they would be lost through predation at the barrier site. Such eels are unlikely to ever contribute to downstream production. Collection elsewhere could result in interbasin transfer of parasites and disease and in extreme cases, reduced recruitment in downstream areas. Removing and increasing access past smaller barriers in streams and rivers may increase eel ingress to farm dams, where fishing is permitted in Queensland. It is also likely to increase the biomass of eels, an important natural predator in east Australian freshwater ecosystems (where it has been reduced below normal levels), and therefore increase total spawner production.

Management at finer scales

The growth rate, the sex ratio, and the future sex ratio of the undifferentiated eels vary between areas at all scales, with differences between latitudes and between habitat types within streams. Individual fishers could maximise their own yield by adjusting fishing pressure and minimum size to the areas in which they fish. The appropriate size and fishing pressure would depend on the species and sex composition of the area. Such management by individuals would be facilitated by giving individuals exclusive access to areas. Competition between individuals in an area would drive them to catch smaller eels and reduce the efficiency of the fishery.

Assumptions and model structure

As with any modelling exercise, there are a number of uncertainties to do with parameter estimates and the type of model used. The spawning per recruit predicted by this modelling exercise is very uncertain, since the current level of fishing mortality is unknown, and highly variable between areas (Chapter 5). Higher exploitation rates would affect spawner numbers more severely.

An important unknown variable is the size selectivity of the fishery. The sampling used to obtain these data was size selective, both due to the methods used (electrofishing and trapping, which are both size selective (Anderson, 1995), and the locations sampled. Larger eels are ecologically dominant (Chisnall and Hicks, 1993), and probably have a larger home range than smaller eels, so are more likely to encounter fishing gear. In addition, since the larger eels are more valuable than smaller eels, fishers preferentially target them. Estimating this selectivity function would improve the predictions of the model, and as discussed above would reduce the estimates of spawning per recruit.

It is difficult to ascertain the influence of eel density on growth, sex differentiation, and maturation, but density is likely to be important in determining both yield and egg production (Horn, 1996; Vollestad and Jonsson, 1988). Sex determination of eels is influenced environmentally and is likely to be affected by density (Colombo and Grandi, 1996; Beullens *et al.*, 1997), with males tending to be more common at higher densities (Krueger and Oliveira, 1999). Size distribution may complicate density effects, since eels change diet as they grow (Ryan, 1986; Jellyman 1989; Sagar and Glova, 1998), and different size classes may not compete with one another. Large piscivorous longfins may reduce the densities of fish that compete with smaller longfins, but they are also cannibalistic (Jellyman, 1989), and are likely to influence the distribution of smaller eels.

The modelling approach I have used is quite simple, and focuses on the objective without incorporating unnecessary or unknown areas. If the basic rule about choosing model complexity is ‘Let the data tell you’ (Hilborn and Mangel 1997), then models of eel population dynamics should in general be simple. Complex models can be difficult to parameterise and calibrate, since both eels and eel fishers can be highly variable between habitats at various spatial scales, and important parameters such as density, catchability, and sex ratio are difficult to estimate. Simple approaches often require fewer data for parameterisation and calibration, less computing power to run, and less knowledge to understand what the model is doing. They can also be more robust and have better predictive power (e.g. Ludwig and Walters, 1985; Punt, 1988). Optimal model size is usually much smaller than intuition dictates (Hilborn and Mangel, 1997).

Conclusion

Fisheries for adult eels can severely reduce the number of migratory eels produced, particularly for long-lived species such as *A. reinhardtii*. For this species, fishing pressure can substantially reduce the number of female migrants. When combined with in-stream barriers, this can severely affect egg production. Egg production can be maintained via unfished areas that have unrestricted access to the sea. This would be best achieved by maintaining the current practise of prohibiting fishing in unimpounded waters, and by increasing upstream and downstream access of glass eels, elvers, and mature females. Management must ensure that sufficient productive areas remain unfished to maintain the supply of recruits to the population. Since there are no reliable estimates of what proportion of unfished egg production is required, conservative management based on the precautionary principle must be applied. Population modelling has the marked advantage of being able to work at such scales, and provide a predictive dimension to the responses of eel populations to changes in management.

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Chapter 7: A management model for eel fisheries

Simon Hoyle

Introduction

Eel life history is unique, and human activities affect their population dynamics differently from other fished species. Recruitment is a serious problem in eel fisheries — it appears to have declined greatly in most eel species in recent years. Managers require advice on ways to maintain eel spawning biomass, but also a tool for understanding how human activities affect eel spawning biomass. An interactive model can help managers gain an intuitive understanding of eels' unusual management characteristics.

In this paper I describe a new model, EELSIM, for interactive modelling of eel management and population dynamics.

Model Description

Here I present a basic overview of the model.

The model is implemented in the widely-used spreadsheet Microsoft Excel. There are three worksheets: 'User', 'Data', and 'Working'. The 'User' worksheet (Figure 1) is the front end, where the user can adjust parameters and read results. The 'Data' worksheet contains the parameter values used in the model for each species/sex or other subgroup modelled. The 'Working' worksheet does the detailed calculations to produce the results that are returned to and displayed by the 'User' worksheet.

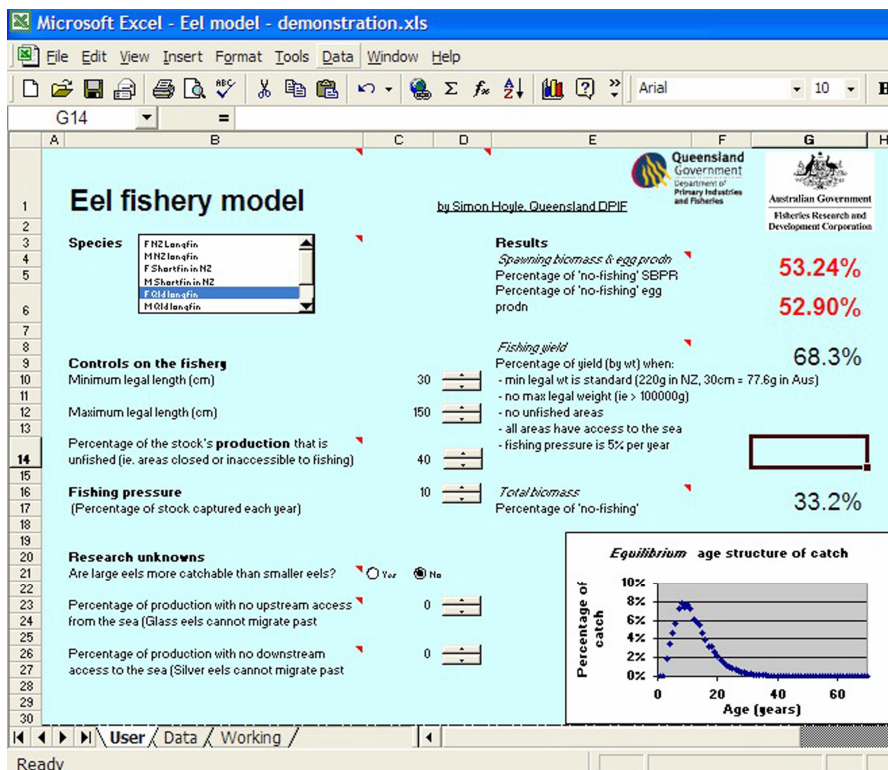


Figure 1: Screen capture of User worksheet.

The model works in terms of production (biomass per area per unit time) rather than area. Comparisons between areas must take into account that, say, lowland pasture is more productive than high altitude forest. User worksheet: Model inputs

Species

Use the list box to choose one of the species/sex/subpopulation options to model. The parameters for each choice are stored in the 'data' worksheet. Additional species/sexes/subpopulations can be added — see the 'Data' worksheet section.

Controls on the fishery

Here the user adjusts the management controls. These are the parameters over which fisheries legislation generally has some control, and which can be adjusted.

Minimum legal weight or length

The legislated lower size limit for adult eel capture fisheries can be adjusted between 0 and 30 000 grams for NZ species or minimum legal size in cm can be selected for Australian species. The lower limit of actual selectivity can also be used. The model uses knife-edged selectivity. More complex patterns of selectivity can be entered by changing the 'Working' sheet. For NZ eels minimum legal sizes in length units need to be transformed into weight using the length-weight relationship.

Maximum legal weight or length

The legislated upper size limit for adult capture fisheries can be adjusted between 0 and 30 000 grams for NZ eels. Similarly maximum legal length in cm can be adjusted for Australian eels.

Percentage of the stock's production that is unfished

Enter a proportion of the production of the fishery that will be permanently unavailable to fishing, between 0 and 100 per cent. These areas may be closed by regulation or simply inaccessible. Percentage of production differs from the percentage of catchment area that is unfished, since all catchments are not equally productive. Unfished areas may be unproductive — for example, alpine national parks in NZ, which cannot be fished, also have very low productivity (Jellyman, 1995). Pasture can be more productive than (generally less accessible) forest (Chisnall and Hicks, 1993).

Fishing pressure

The percentage of fully recruited stock (biomass) captured each year can be adjusted between 0 and 100 per cent. This parameter represents fishing mortality, either uniform across all sizes or increasing linearly with weight. When the parameter 'are large eels more catchable than small eels' is set to 'yes', the fishing pressure parameter represents fishing pressure at 60 cm. See below for more details.

Research unknowns

The model permits the user to specify alternate options for several assumptions.

Are large eels more catchable than smaller eels?

Under the ‘no’ assumption, all size classes are equally catchable. Under the ‘yes’ assumption, catchability increases with size.

According to anecdotal evidence from fishermen, eels become more catchable as they grow larger (Don Jellyman, pers. comm.). This could be caused by eels moving distances in proportion to their size, and so encountering traps and/or fyke nets. It could also be promoted by large eels maintaining a home range and excluding other eels (Broad, Townsend *et al.*, 2002).

Without data to estimate catchability by size in fisheries, the parameters used are hypothetical and represent possible scenarios. They can be adjusted on the 'data' sheet.

In the model including size-dependent catchability, fishing pressure increases at a constant rate with length according to the formula $F_L = F_{base} (1 + (L - 60) \cdot \delta)$, where F_{base} is the fishing pressure parameter and δ is 0.02. Thus fishing pressure is nominal at 60 cm and doubles by 110 cm.

Percentage of production with no upstream access from the sea (Glass eels cannot migrate past barriers)

Dams reduce access to some rivers and creeks for glass eels and elvers. These situations are complex and not well understood. Barriers are often porous to some degree since glass eels can follow water trails up vertical dam walls, and even travel through groundwater. Translocation can also occur. Density-dependence is also very likely to reduce the marginal contribution of each glass eel at higher numbers.

Blocked-off water bodies may in fact continue to yield for many years, as the remaining eels grow. However, this model looks at the equilibrium state, and so ignores the gradual changes (transitional states) between unblocked and blocked production.

Adjust this parameter to define the degree to which equilibrium production of the entire population modelled is reduced by the glass eel blockage.

Percentage of production with no downstream access to the sea (Silver eels cannot migrate past barriers)

Dams block some rivers and creeks to downstream migration by silver eels. These situations are complex and not well understood. Barriers are usually porous to some degree, such that eels may get through at some flows and after some delay. A proportion of migrants may be killed in spillways or turbines etc.

Adjust this parameter to define the proportion of migrating silver eels that are stopped by barriers from successfully reaching the sea.

User worksheet: Model outputs

Spawning biomass — percentage of ‘no-fishing’ spawning biomass

Spawning biomass is given as a percentage of the spawning biomass in a situation where there is no fishing, and all areas have full access to the sea. In other words, with neither fishing nor dams this output would read 100 per cent.

The spawning biomass estimate applies to the fishery overall, and assumes that the fishery is at equilibrium.

Since there is great local variation, and variation through time, this equilibrium figure is unlikely to match observations for a single location, or even a single time averaged over many locations.

Percentage of ‘no fishing’ egg production

This output is the egg equivalent of the spawning biomass output above, derived via the length — fecundity relationship. Larger females tend to devote a greater proportion of biomass to gonad tissue than do smaller females, so increasing their reproductive value. Egg production is a more relevant and important output than spawning biomass, but both are presented because length — fecundity relationships are not available for all eel species.

Fishing yield

Fishing yield is given as a percentage of the fishing yield in a hypothetical baseline situation. This situation is defined for each species, sex and subpopulation in Table .

Table 1: Base levels of fishing mortality and management constraints for each species, sex, and subpopulation.

Location	Species	Sex	Min legal wt	Max legal wt	Per cent unfished areas	Per cent sea access	Fishing pressure
Queensland	<i>A. reinhardtii</i>	F	77.6	30 000	0	0	5%
		M	77.6	30 000	0	0	5%
NSW	<i>A. reinhardtii</i>	F	77.6	30 000	0	0	5%
		M	77.6	30 000	0	0	5%
NZ	<i>A. dieffenbachii</i>	F	220	30 000	0	0	5%
		M	220	30 000	0	0	5%
	<i>A. australis</i>	F	220	30 000	0	0	5%
		M	220	30 000	0	0	5%

The fishing yield estimates apply to the fishery overall, and assumes that the fishery is at equilibrium.

Since there is great local variation, and variation through time, this figure is unlikely to match observations for a single location, or even a single time averaged over many locations.

Total biomass

Percentage of ‘no-fishing’ biomass. No-fishing biomass is the biomass at zero fishing mortality when all productive areas have full access to the sea. This estimate applies to the fishery overall, and that the population is at equilibrium. Since there is no equilibrium, but great variation locally and through time, this figure is unlikely to match observations for a single location, or even a single time averaged over many locations.

Data worksheet

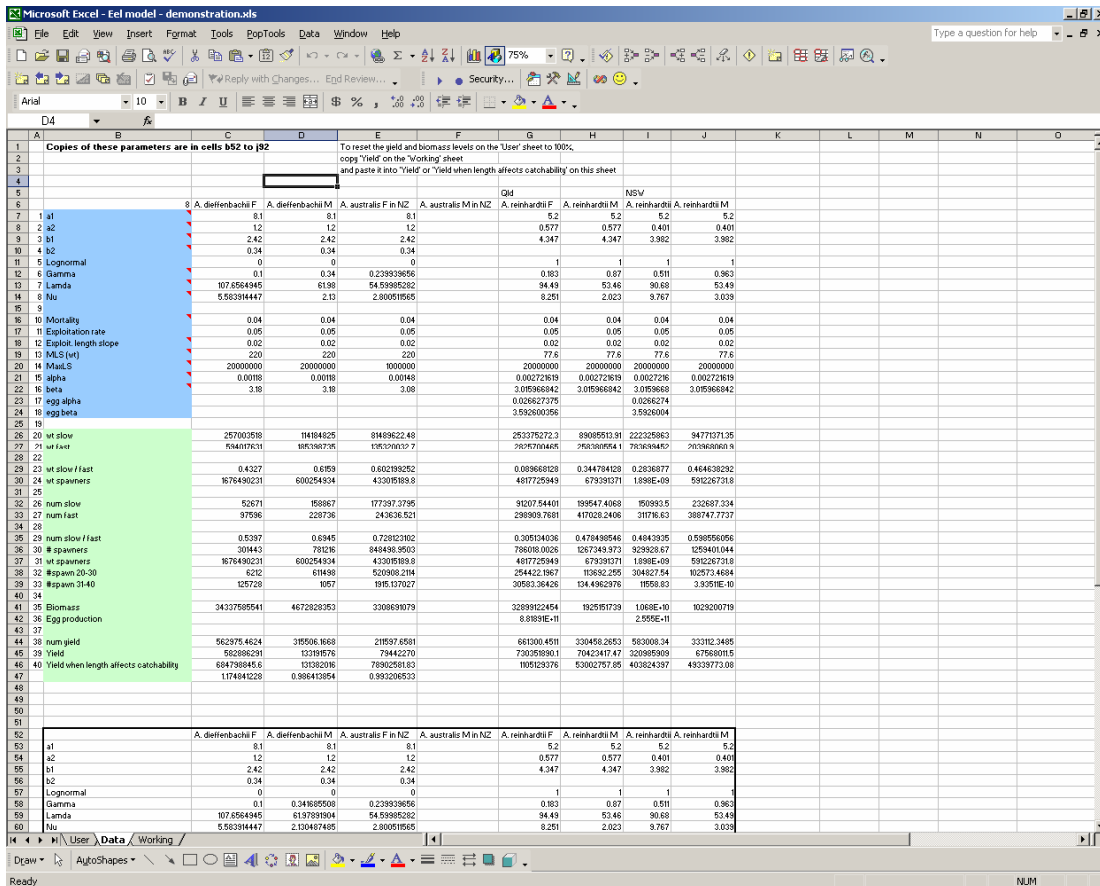


Figure 2: Screen capture of Data worksheet.

The data worksheet (Figure 2) contains model parameter values, which are used in the model in the ‘Working’ worksheet (Figure 3). Each column contains parameter values for a species/sex/subpopulation. The parameter values on this worksheet can be adjusted and columns added for new species and subpopulations.

To add a new species/sex/subpopulation, do the following:

1. Add the data in the column to the right of the last one, in the 18 rows from ‘a1’ to ‘egg beta’.
2. Change the list box on the ‘User’ worksheet
 - a. Add the name of the species/sex/subpopulation to the list in column V.
 - b. Change the input range of the list box to include the new cell in column V
3. Set up unfished biomass and egg production
 - a. Set fishing pressure and the access parameters to zero.
 - b. Copy the results in ‘Working’ worksheet cells B29:B45 and paste the values (using the excel operation ‘paste values’) to cells $\gamma 26:\gamma 42$, where γ is the column holding the new data.

4. Set up yield comparisons

- Set fishing pressure and legal sizes to the desired level for comparison, and 'Are large eels more catchable than small eels' to 'No'.
- Copy the results in 'Working' worksheet cells B47:B48 and paste the values to cells $\gamma44:\gamma45$, where γ is the column holding the new data.
- Set 'Are large eels more catchable than small eels' to 'Yes'.
- Copy the results in 'Working' worksheet cell B48 and paste the values to cell $\gamma46$, where γ is the column holding the new data.

If any of the parameter values are changed for an existing species/sex/subpopulation, steps 3 and 4 should be carried out to reset unfished biomass and yield comparisons.

Working worksheet

The screenshot shows a Microsoft Excel spreadsheet titled 'Eel model - demonstration.xls'. The active cell is B47, containing the formula '=AG134*AG46'. The spreadsheet is organized into columns labeled A through X and rows numbered 1 through 64. The data includes various parameters such as 'Age', 'Length', 'Weight', 'Mortality', 'Exploitation rate', 'Fishing slope', 'M.L.S (wt)', 'M.L.S (len)', 'alpha', 'beta', 'egg alpha', 'egg beta', and 'Biomass (relative)'. The bottom of the spreadsheet shows a 'User' field with the name 'Working' and a 'Ready' status bar.

Figure 3: Screen capture of Working worksheet.

The structure of the model contained in this worksheet is described in detail in Hoyle (2004). The basic model processes and the order in which they occur are shown in Figure .

Note that on the right hand side of the worksheet (columns DE to EM) are a number of tables which supply data for figures presented in Hoyle (2004), some removed but some remaining in the spreadsheet. These are left as examples for users to copy and paste into their own data.

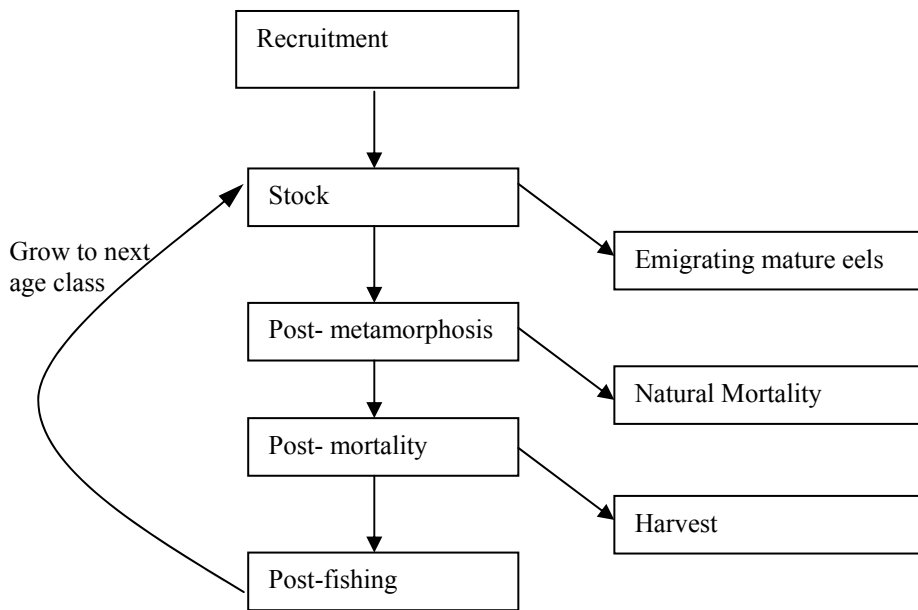


Figure 4: Schematic diagram of model processes. Each cycle through the model takes one year.

Model Features and Conclusion

Examples of application

Two examples of application of this model are (Hoyle and Jellyman, 2002), and Hoyle 2004 (this volume). The first describes results for *Anguilla dieffenbachii* and *A. australis*, the two species in the New Zealand eel fishery. The second describes results for *A. reinhardtii*, the Australian longfin eel, in Queensland and New South Wales.

Major limiting assumptions

The major limiting assumption of the model is the fact that yield per recruit is an equilibrium approach. Equilibrium models ignore important dynamic features of biological systems, which can significantly affect outcomes. However, eel populations are so greatly subdivided (Dekker, 2000), with spatial and temporal variability in most parameters occurring on small scales, that synthesis using equilibrium models is likely to be valid. One parameter likely to vary on a larger spatial scale — recruitment — is not included in the model, since yield per recruit only considers individuals once recruited.

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Chapter 8: Implications for Australian longfinned eel management

Michael Hutchison, Wayne Sumpton and Simon Hoyle

Abstract

The current eel fishery management systems in Queensland and NSW essentially follow the principle of eel reserves, thus in theory these fisheries should be sustainable provided the area of reserves are adequate. Escapement of spawners is probably adequate but sustainability of impoundment fisheries could be at risk through inadequate recruitment back into the impoundments. Barriers to migration are a major risk to the eel fishery. Likely solutions to this problem include construction of elver passes or trapping, transport and release of glass eels and elvers from downstream areas into impounded waters as per the Victorian shortfinned eel fishery. There is potential to increase the carrying capacity of both eel reserves and impounded waters through habitat remediation and enhancement.

Project Objectives

1. Develop a management population dynamics model and use it to investigate management options.
2. Establish baseline data and sustainability indicators for long-term monitoring.
3. Assess the applicability of the above techniques to other eel fisheries in Australia, in collaboration with NSW.

Introduction

This chapter draws on the results of the previous chapters as well as existing scientific literature and data sets to evaluate implications for the management of eel fisheries in NSW and Queensland. The focus of this document is not solely on management of the fisheries themselves, but also on management of the habitats that sustain these fisheries.

Current management arrangements in NSW and Queensland and implications of the eel fishery model for eel fisheries management

As outlined in Chapter 1, in Queensland the commercial adult eel fishery is a trap fishery and is restricted to freshwaters impounded by an artificial human made structure (e.g. a dam weir or barrage). This definition includes farm dams (Zeller and Beumer, 1996). The fishery is further restricted by a prescribed number of traps and weekend closures (Fisheries Freshwater Management Plan, 1999 reprint No. 2B, 2003). There is no total allowable catch restriction due to the current small number (26) of licensed trappers (Wohlsen *et al.*, 2002).

The NSW eel fishery is also a trap fishery and has approximately 200 eel fishers endorsed to fish in estuaries. About 13 fishers currently fish impoundments and farm dams. Non-tidal riverine areas are closed to commercial harvest (Pease, 2000). No new fishers have been issued permits since 1992 (Pease and Walford, in prep.). Both the NSW and Queensland fisheries have a minimum legal size of 30 cm total length for commercially and recreationally harvested eels.

The minimum legal size of 30 cm is well below the size at 50 per cent maturation for both male (530 mm) and female (955 mm) longfinned eels. With maturation times recorded in Queensland between 4 and 17 years for males and 5–39 years for females in Queensland, and 5–22 years for males and 5–52 years for females in NSW (Walsh *et al.*, 2004) (there is a considerable window of

opportunity for fisheries to capture eels prior to maturation and migration to spawning grounds. This is why the sustainable management method suggested by the fishery model is for eel reserves. In other words closing catchments or regions to commercial harvest, to enable a proportion of the total eel population to mature and migrate.

In effect both the Queensland and NSW management systems have unimpounded freshwaters acting as eel reserves. Therefore the current management system is in agreement with the recommendation derived from the eel fishery model, i.e. to have eel reserves to ensure adequate spawner escapement for long-term sustainability of eel stocks. Exploitation of yellow eels, **without closed areas**, at rates of 5 per cent and 10 per cent per year reduced the egg production per recruit of longfin females by 57 per cent and 80 per cent respectively below unexploited levels.

At first glance it may appear that with the NSW eels fisheries being concentrated in estuaries that they have the potential to capture a large proportion of migrating silver eels as they leave freshwaters and pass through estuaries on their way to the Pacific Ocean, thereby negating the effect of the eel reserves.

If the NSW estuary fishery relied on fykes or other unbaited winged traps that gather fish as they move along river or estuarine channels, then there could well be a risk of low escapement of spawners. However this is not the case. The NSW estuarine fisheries are based on a baited trap fishery (the traps are not winged) and depend on eels moving into the trap to feed. Migrating silver eels are unlikely to feed and therefore the proportion of migrating silver eels likely to be caught is low.

Sustainability of eel stocks in Queensland and NSW are therefore unlikely to be adversely impacted by the current fishery arrangements. However there could be potential problems for the sustainability of the fisheries themselves within some impounded waters. This problem will not be due to insufficient spawning stock, but rather through lack of, or insufficient recruitment back into impounded waters after they have been fished down.

Effect of changing legal sizes

Modelling in Chapter 6 suggested that the best option for sustaining eel stocks was to close areas to commercial harvest and to manage them as areas for spawner production. The closed areas in Queensland consist of both estuaries and unimpounded freshwater riverine habitats. In NSW closed areas consist of unimpounded freshwater reaches only. Male longfinned eels tend to be concentrated in the lower freshwater reaches and in estuarine areas.

Thus, freshwater reserves tend to protect females better than males. The ratio of males to females appears to be lower in NSW than in Queensland (Chapter 5 this report and Walsh *et al.*, 2003). NSW Fisheries (DPI) have expressed some concern about this situation. If eel populations are panmictic, then this is probably not a problem, as male eels from Queensland should more than compensate. Modelling in Chapter 6 also suggested that the selectivity of the fishery due to market reasons was permitting considerable escapement of males.

Fishing pressure is unlikely to be significantly affecting male spawning biomass. However, Bruce Pease (NSW Fisheries/DPI&F) has expressed concern that some fishers are targeting smaller eels in NSW estuaries. Increasing the minimum legal size for eels in NSW would ensure the majority of male eels escape the fishery to spawn.

For female eels the optimum yield-per-recruit at the current minimum legal size (30 cm) was obtained with a moderate exploitation rate of between 10 per cent and 15 per cent. However, yield at these exploitation levels was increased by raising the minimum legal size, which reflects the actual selectivity of the New South Wales fishery. A large proportion of this selectivity is achieved by voluntary release of eels smaller than 500g (58 cm mean length) not the mesh size of the traps (Pease* pers. comm.). The optimum minimum legal size increased as the exploitation rate increased. In theory, raising the minimum legal size would increase yield of the female fishery, which on the face of it may seem an additional benefit, but in practice it will make little difference to the current situation as current selectivity, based on market demand is already around 58 cm.

In summary changing the minimum legal size is probably not necessary as a sustainability measure, given the current management practice of closed areas to fishing, and current levels of selectivity and escapement of males. However, given the concerns of NSW fisheries about low numbers of males in that state, increasing the minimum legal size to above 50 cm could be used as a precautionary measure.

In Queensland, given closure of estuaries and freshwater unimpounded reaches to commercial harvest of eels and effective size selectivity at 58 cm, changes to the minimum legal size cannot be justified.

Declining impoundment fisheries, barriers to migration and other habitat issues

The CPUE of the Queensland longfinned eel fishery has begun to decline (Wohlsen *et al.*, 2002). In contrast the NSW fishery, which is dominated by estuarine catches has had relatively stable catches since the mid-1990s (Pease and Walford, in prep). In Queensland there is already evidence of very few eels upstream of Wivenhoe Dam (EHMP, 2004, and Chapter 1 this report) and most of these are old large eels. No commercial catches have been logged from Lake Wivenhoe since 1997. Commercial catches declined from 30 tonnes in 1995–96 season to 10 tonnes in the 1996–97 season, and 1.7 tonnes during the major catching period of the 1997–98 season. We even caught no eels in Tingalpa reservoir, which has not been fished commercially, although one of us (MH) has subsequently observed eels in this waterbody. Loss of eels from Tingalpa reservoir could be through downstream migration of eels during overflows, but with little recruitment upstream of the barrier. At the Ben Anderson Barrage near Bundaberg, Stuart and Berghuis (1999) noted longfinned elvers were absent from the top of the vertical slot fishway, but were abundant at the bottom. Elvers were also observed congregating on the concrete weir crest beneath leaks in the collapsible shutters. These authors recommended installation of an elver pass to assist passage of elvers and glass eels past this barrier.

Although eels are capable of migrating around barriers when conditions are damp and although glass eels can migrate up damp surfaces on weir walls, opportunities for migration become more limited with increasing height of the structure and risk of predation is increased when glass eels and elvers are forced to leave the water. Higher weirs with less frequent overflow can be expected to have reduced recruitment rates. Figures 1 and 2 show examples of CPUE declines in two impounded Queensland waters above high weirs.

* Bruce Pease, NSW Fisheries. Principal Investigator FRDC Project 98/127.

The eel fishery in Queensland was initially set up as a predator control fishery, with the aim of removing eels from impoundments stocked for recreational fishery enhancement purposes. It was thought that eels would impede the success of fish stocking programs by preying on stocked fingerlings. It would appear that in some impounded waters the original goal of eel removal has been achieved. However, the original motive for eel removal seems flawed. Hutchison *et al.*, (2004) found little evidence to support eels having a major impact on non-translocated species stocked in coastal impoundments, although eels may have some impact on stockings of translocated silver and golden perch. The ecological implications of removing a top native predator and scavenger were also not considered. It would be preferable both for the ecosystem and the eel fishery if impoundment eel fisheries were sustainable and eels returned to impoundments. The stocked impoundment fisheries program would probably fail EA sustainability criteria if elimination of eels from impoundments was maintained as part of the program.

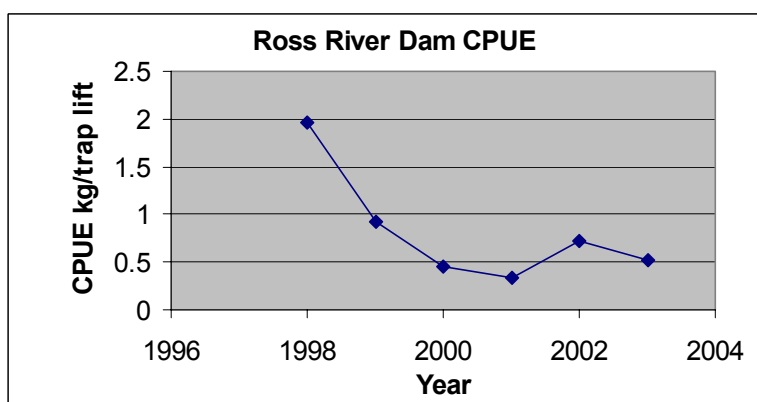


Figure 1: CPUE kg/trap lift of longfinned eels in Ross River Dam (Source data C fish data base QFS).

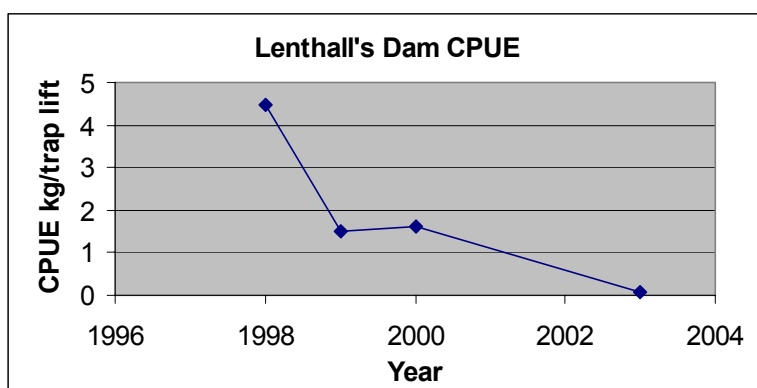


Figure 2: CPUE kg/trap lift of longfinned eels in Lenthall's Dam (Source data C Fish database QFS).

The current management arrangements for longfinned eel fisheries in Queensland and NSW basically follows the eel reserve management methodology suggested by the eel population model. Given their life history, eels can be recruitment overfished without being growth overfished. It is unclear what proportion of the unfished egg production will maintain recruitment at replacement levels. This question has been addressed for a range of stocks with the conclusion that, where there is no basis for estimating the replacement level of spawning per recruit, 30 per cent of the unfished

level is a conservative default value (Mace and Sissenwine, 1993). Given the particular vulnerability of long-lived species, and the uncertainty about the current level of longfinned eel depletion, a more conservative level such as 50 per cent of unfished spawning per recruit may be justified (suggested by Jellyman, 1993 with respect to escapement of migrating longfinned females in New Zealand). One way of ensuring the unfished stock is above 50 per cent is to ensure passage of eels past barriers.

Assuming there is sufficient spawner escapement from unfished unimpounded waters, in theory the impoundment and farm dam fisheries can be managed sustainably in both states provided recruitment back into the impounded waters can be ensured. In impounded waters with low barriers (e.g. some farm dams and small weirs) natural recruitment should continue fairly regularly. Graynoth and Niven (2004 in press) assumed that dams less than 3 m high have little effect on movements of New Zealand longfinned eels. However, in waters above high weirs or with weirs that spill infrequently sufficient recruitment to sustain the fishery is likely to be a problem and fisheries for eels are likely to decline. Recruitment problems are likely to be compounded in waters in which there are a series of weirs, with recruitment being progressively reduced in an upstream direction. The Kolan river (Figure 3) is one example of such a river.

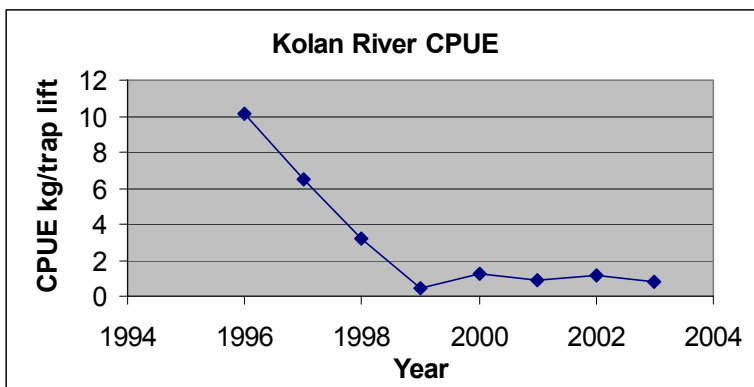


Figure 3: CPUE of longfinned eels in the Kolan River system.
This river has a series of weirs. (Source data C fish data base QFS)

There are well over 3000 weirs dams and levees documented blocking water courses in the east coastal catchments of Queensland (Source DNR&M and QFS databases). Although many of these would only be minor obstacles for eels, approximately 40 per cent are greater than five metres in height. Pethebridge *et al.* (1998) documented 254 obstructions to fish passage in south coastal streams of NSW, including high dams, fixed crest weirs and tidal flood gates, culverts and causeways. The total number of barriers in all coastal streams in NSW would be much higher than this figure. Some of these barriers can be expected to be significant obstacles to upstream recruitment by eels.

Barriers to eel migration have also been considered problematic in other parts of the world (White and Knights, 1997; Allibone, 1999), and some regions already have installed elver passes or participate in trap and transport (Knights and White, 1998; De Leo and Gatto, 2001).

Potential solutions to the problem of barriers to upstream recruitment include provision of elver passes (e.g. Stuart and Berghuis, 1999), or trapping below major barriers, followed by transport and release of glass eels and elvers into impoundments above major barriers. Initially releases could be concentrated on impounded waters where major declines in the eel fishery have been documented (e.g. Wivenhoe Dam). Further research is required into elver passes and obstacles to eel migration

in Australia, but this is an area of research that should be pursued. Barriers to migration are certainly an issue in shortfinned eel fisheries in Victoria. Trapping and transport is already carried out in Victoria, where permit holders are licensed to release juvenile shortfinned eels into crown waters for on-growing under natural conditions (McKinnon, 2002). Such a practice in NSW and Queensland would help ensure the ongoing sustainability of eel fisheries in impounded waters and would have the ecological benefit of returning a predatory fish to its natural range. There should not be any genetic problems with such a practice, as the stocked eels would be sourced from wild stock attempting to migrate into the waters upstream of their point of capture.

Long-term monitoring and managing habitat for eels

In Chapter 1 it was established that the QFS long term monitoring program will be suitable for detecting changes in the abundance of eels greater than 300 mm total length. CPUE effort data from the long term monitoring program can therefore be used as an index of abundance. Should the QFS long-term monitoring program detect cross catchment declines in eel stocks, then this may indicate that unfished waters are insufficient to maintain an adequate eel spawning stock, in which case the number of reserved waters or catchments may need to be re-evaluated in both Queensland and NSW. On the other hand if declines are detected in some catchments but not others, this could indicate local environmental effects impacting on recruitment or survival of eels. Cumulatively such local effects may eventually lead to more global impacts if the total number of spawners is significantly impacted, such that production of glass eels declines to the point where density dependent recruitment effects take over. Any sustained decline (say over three consecutive years) at a level greater than normal interannual variance should trigger an investigation into possible causes and review of management of the fishery.

The QFS long-term monitoring survey data and the SEQ eels survey data (Chapter 1) has given some indication of habitats likely to be favoured by eels. For example small eels were shown to favour shallow riffle and run type habitat and rocky substrates. Large eels were likewise associated with rocky habitat and with undercut banks and large woody debris. Using this knowledge it should then be possible to rehabilitate streams and rivers to make them into more favourable habitats for eels. Such works should be undertaken in both fished and unfished catchments thereby maximising the utility of eel reserves and the effectiveness of eel stocking programs above barriers.

In our surveys we did not capture any eels smaller than 300 mm in impounded waters. This suggests impounded waters may not favour small eels. Therefore it may be necessary to create complex rocky habitats within impounded waters, or to create or restore rocky riffle habitats in stream sections upstream from impoundments in order for juvenile eels to recruit to the impoundment fisheries.

The aim of habitat restoration within unfished systems should be to increase the carrying capacity of these systems thus increase the number of spawners that migrate back to the ocean. A GIS system as outlined in Chapter 1 could initially be used to select those catchments most in need of restoration and most likely to be occupied by eels following restoration. Evaluation of habitat works should be undertaken to see if they are having the desired impacts. Potential habitat works that could be undertaken in such systems could include resnagging with logs, creation of artificial undercut banks through strategic placement of logs or use of devices such as L.U.N.K.E.R.S (Brown, 2003) and restoration or creation of rocky riffle habitat (Torre, 2001) for juvenile eels. In many catchments these types of habitat have been lost through bank slumping, channelisation, downcutting and siltation and through deliberate removal of instream large woody debris. Even though such restoration works will be directed at sustaining eel populations they should also have a benefit to other species that rely on rocky riffle, large woody debris and undercut bank habitats.

A number of manuals are now available on stream restoration works, including use of large woody debris, rock rip-rap, riffle creation and construction of undercut banks (e.g. Newbury and Gaboury, 1994, Torre, 2001, Rutherford *et al.*, 2000).

Recommendations

1. Continue to exclude unimpounded freshwaters from the commercial eel fishery to ensure adequate spawner escapement and sustainability of the fishery as per the eel reserve model.
2. As a precautionary measure to provide increased protection for male eel stocks in NSW increase the minimum legal size to above 50 cm in that state. No change in legal size is required in Queensland.
3. Conduct research into elver passes and introduce their use on barriers to eel migration throughout Australia.
4. Introduce the practice of trap and transport of elvers and glass eels around major barriers to sustain recruitment of eels into impounded waters until effective elver passes are implemented.
5. Monitor riverine eel stocks through the long term monitoring program. Use the long term monitoring CPUE data as an index of abundance. Investigate probable causes of catchment specific declines or statewide declines should any sustained drop in CPUE substantially greater than normal interannual variance be detected over three consecutive years.
6. Use habitat remediation to favour recruitment and survival of eel stocks both in waters upstream of impoundments and in unimpounded reserved waters.

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Chapter 9: Benefits and adoption

Michael Hutchison and Simon Hoyle

Benefits

Research results from this program will benefit the long-term sustainability of eel fisheries. The results of this research will therefore be of benefit and value to both eel fishers and fisheries managers.

Fisheries in both NSW and Queensland already are run along the lines of having substantial areas closed to the adult eel fishery. Within Queensland, riverine and estuarine areas are closed to adult eel harvest and fisheries are confined to impounded waters. Within NSW the fishery is excluded from freshwater riverine areas and focused on estuaries, with some additional activity in some impounded waters. Enough spawners probably escape to ensure ongoing unexploited stocks of eels are sustained. Management changes required are therefore minimal.

Modelling has suggested that yield can be increased theoretically by increasing the minimum legal size, however market forces have in effect already imposed size selectivity at above 50 cm, therefore increasing minimum legal size is unlikely to have a great impact on yield. Increasing the minimum legal size will also increase escapement of male eels, which is an outcome desired by NSW DPI, to prevent targeting of male eels for aquaculture. Therefore, in NSW, it is proposed to increase the minimum legal size.

However, increasing escapement will probably not have any major effect on sustainability, as current selectivity of the fishery and closure of Queensland Estuaries already ensures most males escape to spawn. With the current system of closures, increasing the minimum legal size is not really essential for sustainability.

Data from this study made a substantial contribution to a report to Environment Australia (now DEH), (Wohlsen *et al.*, 2002) which enabled the Queensland fishery to obtain an export license as a sustainably managed fishery. Results from this study have also contributed to recent negotiations between DEH and DPI&F.

Current exploitation levels of glass eel stocks appear to be at a level that is below that where density dependent effects begin to impact on recruitment. A recent introduction of a quota on glass eel harvests in Queensland and a quota in NSW is likely to ensure this trend continues. Current production of eels would appear limited by the carrying capacity of the available environment. However, much of the past freshwater environment of eels is now inaccessible or partially inaccessible to new recruits.

Through removing barriers, constructing elver passes (current vertical slot fishway designs do not seem to pass glass eels) or trapping below barriers and transporting elvers and glass eels above barriers, more habitat will be opened up to eels and total production of spawners will be increased. Increasing passage of glass eels past artificial barriers will also improve recruitment to the public impoundment fisheries and upstream farm dams. Increased spawner production will ensure the ongoing viability of estuarine fisheries in NSW and impoundment fisheries in both Queensland and NSW.

Adoption

As already stated, minimum changes are required to the fisheries in NSW and Queensland. NSW is considering increasing the minimum legal size of eels. Modelling suggested this will increase yield to the fishery, but the actual increase may be minimal, as most fishers are tending to concentrate their efforts on larger eels already. There will be an increased escapement of males by increasing the minimal legal size, to the extent that most males would be unfished, which may have some benefits to sustainability, but escapement of females from reserved or unfished areas is probably more important to sustainability, as it is predominantly females that are exploited. Based on the recommendations of this study, it is unlikely that the minimum legal size for eels will be altered in Queensland.

Partly as a result of this study, the freshwater long term monitoring program is continuing in Queensland and it has been agreed with DEH that CPUE data from this program will be used as an index of abundance of eel stocks. This will continue to provide data on the abundance of unexploited eel stocks, the potential future spawners. Sustained declines in eel stocks across all surveyed east coastal catchment would provide early warning of potential recruitment problems and could trigger a further review of the eel fishery.

Recommendations on the need for elver passes or trap and transport around barriers have only just been made. There has not yet been sufficient time for fisheries managers in NSW and Queensland to consider this recommendation in depth, however some provision for a cost benefit review of fish passage for eels and trap and transport has been made in an agreement between DEH and DPI&F for management of the eel fishery. Both NSW and Queensland already have fish passage programs in place, and some experiments with elver passes have already been conducted in Bundaberg. It is likely that the results of our study will trigger further action with regard to evaluation of elver passes.

In New Zealand the modelling component of this study has triggered GIS studies to examine the adequacy of existing reserves (Graynoth and Niven, 2004). New Zealand managers seem to be taking the need for reserves seriously, although there is some resistance from industry and an ongoing debate in that country. Modelling on New Zealand eel stocks, by Hoyle and Jellyman (2002) that resulted from our current study, was cited in the Quebec declaration on world eel stocks.

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Chapter 10: Further Developments

Michael Hutchison and Simon Hoyle

With current management practices for eel fisheries in NSW and Queensland largely complying with the key recommendation of this report (management by closures) and with the commitment by the Queensland Fisheries managers to continue the long term monitoring program for eels, little is required in the area of further development, except perhaps for advancement of upstream passage of elvers and glass eels and habitat remediation.

Certainly DEH is aware of the need for elver passes and/or trap and transport to increase spawner production, and this is reflected in an agreement with DPI&F. Potential benefits to the fishery of elver passes or trap and transport are also acknowledged. To further develop this, it is proposed that the results of this work be passed on directly to fish passage researchers in NSW and Queensland, and to their relevant managers, as this may assist in progressing development of elver passes. Copies of the report will also be forwarded to commercial eel fishers and managers of the eel fishery. It is possible that this could lead to some agreements for trap and transport arrangements. Electronic copies of the eel report and model will also be forwarded to ANZERG, and it is possible this group may try to facilitate work on elver passes.

It is proposed to put this eel report and the eel management model on the DPI&F web site. Links to the model and report will be e-mailed to fisheries agencies in Australia, New Zealand, Japan, Europe and North America. It is hoped that the model may contribute to management of eel fisheries worldwide and that progress on effective mechanisms to aid upstream passage of elvers and glass eels can be progressed.

We recommend further research into the habitat requirements of elvers and yellow eels and into what constitutes a productive environment for eels. Some work along these lines has already been undertaken in New Zealand. In New Zealand work on understanding the proportion of eel stocks likely to be protected by reserves has been investigated in part of the South Island using GIS modelling. With improved knowledge of productive eel habitats, similar modelling could be carried out in Australia, to improve our knowledge of the adequacy of reserve systems. It may also be possible to improve recruitment and survival of eels through habitat remediation works. Under NHTII and NAP this is an expanding area of research in Australia. A number of regional NRM groups formed to develop regional strategies for NHTII funding have already included fish passage and habitat remediation in their key priorities. Forwarding a summary of the key findings and links to the model and final report to these groups may also facilitate actions that will benefit eel stocks.

Chapter 11: Planned Outcomes

Michael Hutchison and Simon Hoyle

The objectives of this research project were as follows:

1. Estimate population parameters required for a management model. These include survival, density, age structure, growth, age and size at maturity and at recruitment to the adult eel fishery. Estimate their variability among individuals in a range of habitats.
2. Develop a management population dynamics model and use it to investigate management options.
3. Establish baseline data and sustainability indicators for long-term monitoring.
4. Assess the applicability of the above techniques to other eel fisheries in Australia, in collaboration with NSW. Distribute developed tools via the Australia and New Zealand Eel Reference Group.

The ageing of eels in subtropical Queensland was validated in this study, permitting age structure, growth and age at maturity data to be collected from a wide range of sites in Queensland, enabling parameter estimation to be developed for a Queensland eel fisheries model. Data supplied by NSW Fisheries also permitted parameter estimation and model development for NSW stocks. Density estimation was somewhat less successful, with mark recapture estimates only working for short term within stream experiments. Longer term, emigration, possible drought related mortality made estimation of density in some farm dam sites more difficult. Lack of precise density estimate data did not preclude development of an eel management model.

CPUE data was found to be sufficient for comparing relative abundance of eel stocks between sites and over time. Lack of recruitment into large impoundments (possibly compounded by drought) and past fishing pressure resulted in very low catches of eels in large impoundments.

A management population dynamics was successfully developed during this project. The model has shown that spatial closures and enhancement of passage of eels past barriers are the most effective management options for the eel fishery. Due to the late maturation of eels, minimum sizes were found to be a less effective sustainability measure, especially for female eels. Both Queensland and NSW fisheries already use closures as a management tool in the eel fishery.

Catch per unit effort data from the Queensland Freshwater Fisheries long-term monitoring program (LTMP) was shown to be a suitable sustainability indicator for the longfinned eel fishery. This data also can be used to show influences of habitat on abundance of eels. The South-east Queensland eel surveys used in this study were also shown to be an effective means of measuring abundance of eels for use as a sustainability indicator. However these surveys are more intensive and costly than the current LTMP, and it was recommended that they only be initiated if LTMP was detecting a decline in the fishery.

The eel model has been applied to the NSW eel fishery, and came up with similar results as those for the Queensland eel fishery. Data from the New Zealand eel fishery has also been applied to New Zealand longfinned eel and shortfinned eel stocks. This has suggested that New Zealand longfinned eel stocks are more dependent on reserves than New Zealand shortfinned stocks. Should data be provided on Australian short-finned eel stocks, then this model could also be successfully applied to

them, and be used to investigate management options such as changing harvest rates, minimum and maximum legal sizes, use of closures and provision of passage past barriers to upstream migration.

Developed tools have not yet been distributed to the Australian and New Zealand Eel Reference Group, at the time of writing, but will be following finalization of this report.

In the old project proposal format in which this project was submitted there was no section specifically identified as planned outcomes or outputs. Nevertheless the intention of the project was to lead to sustainable management of long-finned eel fisheries, with the primary output of the eel management model informing this outcome.

To a large extent, existing management of eel fisheries in NSW and Queensland match this desired outcome. The results of this study have been able to reinforce that the fisheries are being managed appropriately in both states. Therefore the current system of closures will remain. DEH and DPI&F have already reached an agreement on how to proceed with management of eel fisheries. Most of the data required for the development of this agreement was derived from the research in this project and the management models outputs. Development of adequate passage for elvers past barriers such as weirs and dams is still required as part of the sustainable management of long-finned eels fisheries outcome, but the issue has been recognised by DEH and DPI&F and it is likely to progress in coming years.

Chapter 12: Conclusion

Michael Hutchison and Simon Hoyle

The objectives of this project have been substantially achieved. Parameters such as growth, age, size and age at maturity, size selectivity by the fishery have been estimated and contributed to the development of an eel fishery management model. This has already had an influence on an agreement between DEH and DPI&F on how to manage long-finned eel fisheries in Queensland. The project has also been able to establish that the catch per unit effort data from the DPI&F freshwater fishery long term monitoring program is a suitable index of abundance to use as an indicator of sustainability. The LTMP is able to detect between catchment and between year variation in CPUE and is also able to account for environmental effects on abundance.

The eel fishery model developed during this project is applicable to other eel fisheries in Australia and around the world. Once parameters have been identified or supplied for other fisheries, these can be incorporated into the model, which can be used as a management tool for those eel fisheries. The model already incorporates data for Queensland and NSW long-finned eel fisheries and for New Zealand short-finned and longfinned eel fisheries. The model has already had an impact in New Zealand, where the issue of eel fishery closures is being debated and where further research into adequacy of closed areas has been initiated.

In conclusion, female long finned eels in Queensland mature between the ages of 5 and 39 years, and males mature between 4 and 17 years. Growth is highly variable between locations. Given the long maturation time of eels and the fact that they only spawn once, long-finned eels are vulnerable to over-exploitation. Using measures such as minimum size or catch quotas to manage their stocks is not sufficient on its own. Exploitation of yellow eels, without closed areas, at rates of 5 per cent and 10 per cent per year reduced the egg production per recruit of long-finned females by 57 per cent and 80 per cent respectively below unexploited levels. The model suggests that, within a population subunit, management of Australian longfinned eels for egg production conflicts with management for fishing yield. Access of glass eels during upstream migration is also likely to reduce egg production across large areas. Separation into areas for egg production (areas closed to fishing) and areas for fishing yield, combined with restocking or improvements in eel access into catchments with significant barriers to migration, is likely to achieve better outcomes for both. The current eel fishery management methods used in both NSW and Queensland are essentially along the lines suggested by the model, although more needs to be done to improve recruitment of eels into areas upstream of barriers.

Appendix 1

Intellectual property

There are no intellectual property issues arising from this project. The eel model software is non-commercial and can be accessed freely on the internet and is provided in a CD ROM with hard copies of this report.

Appendix 2

List of staff involved on the project

Simon Hoyle	Principal Investigator, Fisheries modeller
Michael Hutchison	Senior Fisheries Biologist
Michelle Sellin	Fisheries technician
David Peel	Fisheries modeller
David Mayer	Biometrician
Wayne Sumpton	Fisheries Biologist