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Optimal management of the New Zealand longfin eel (*Anguilla dieffenbachii*)*

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Annual recruitment of the New Zealand longfin eel (*Anguilla dieffenbachii*) has decreased by 75 per cent since significant levels of commercial fishing began in the early 1970s. This motivates application of a multiple-cohort bioeconomic model to a New Zealand longfin eel fishery to investigate its optimal management and ascertain the suitability of existing regulatory policy. The use of historical harvest to calculate total allowable catch is asserted to be unsustainable based on recovery dynamics. In addition, individual transferable quota systems are argued to be fundamentally flawed for the protection of longfin fisheries because of high-grading, low-surplus production and a current lack of effective stock-assessment procedures. Area closure and the spatial definition of harvest rights are attractive alternatives given the territoriality of longfins and high larval spillover. The importance of unfished reserves is reinforced when significant uncertainties regarding population strength, harvest intensity and growth dynamics are considered. Restriction of exploitation to older cohorts in fished areas is demonstrated to maximise economic yield.

Key words: *Anguilla dieffenbachii*, area closure, fishery management, longfin eel, territorial user rights.

1. Introduction

Freshwater eel populations (*Anguilla* spp.) throughout the world are extremely sensitive to exploitation. Annual harvesting has a cumulative impact on year-class strength given that eels take many years to reach sexual maturity. Moreover, harvested eels have never spawned because anguillids are semelparous, dying after making a single reproductive contribution. Together with habitat loss and physical barriers to migration (Castonguay *et al.* 1994), overexploitation is estimated to have decreased annual recruitment of the European eel (*Anguilla anguilla*) by 99 per cent (International Eel Symposium 2003). Similarly, recruitment of the American eel (*Anguilla rostrata*) into the Saint Lawrence River is estimated to have declined by over 95 per cent between 1982 and 1995 (Casselman *et al.* 1997). The decline of the world's longest living anguillid, the New Zealand longfin eel (*A. dieffenbachii*), is consistent with these Northern

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Hemisphere examples. Biologists have estimated that annual recruitment of this species has declined by 75 per cent since the start of significant commercial exploitation in the early 1970s (Jellyman *et al.* 2000).

Overexploitation of the longfin resource threatens the continued provision of a number of important values. A considerable proportion of annual harvest (35–40 per cent) in the New Zealand commercial eel fishery consists of longfins. The remainder are either earlier maturing shortfins (*A. australis*) or the seldom-encountered Australian longfin (*A. reinhardtii*). This industry has historically been very significant, with freshwater eels being the second-most valuable fish export from New Zealand in 1975. Although this industry is now much smaller, it contributes approximately $NZ15m (including economic multiplier effects) annually to the New Zealand economy. The longfin also supports a significant customary fishery given the traditional harvest rights of New Zealand's indigenous people, the Maori. The New Zealand Ministry of Fisheries is obligated to protect these rights under the 1840 Treaty of Waitangi. There is also a small recreational fishery for this species.

Overexploitation of the longfin fishery has been promoted through a number of factors. Open access conditions have prevailed in this fishery for most of its history. The resultant competition among commercial, customary and recreational fishers has severely affected spawning biomass (Hoyle and Jellyman 2002). Prices for eel products have also remained consistently high because of significant global demand. This has encouraged a greater fishing effort because harvesting costs are low in comparison. Furthermore, weight and gear restrictions have failed to protect spawning biomass (Hoyle and Jellyman 2002; Chisnall *et al.* 2003), and effective limits on effort or harvest did not exist for many years. A step towards potential improvement in the management of this fishery is the recent adoption of individual transferable quotas (ITQs) (Grafton 1996). These may provide an incentive for conservation through the allocation of property rights; however, a lack of biological data has prevented accurate stock assessment. This has led to the use of historical harvest figures, consistent with severe recruitment overfishing, to calculate total allowable catch (TAC) levels.

This paper focuses on the investigation of sustainable management strategies for the New Zealand longfin eel fishery. It utilises a bioeconomic model based on the general framework of Conrad (1982). This contribution is valuable because existing research into the management of this fishery (Jellyman *et al.* 2000; Hoyle and Jellyman 2002) excludes both economic and dynamic effects. Existing weight and harvest restrictions are shown to be inefficient and detrimental to the long-term viability of longfin populations. Uncertainty surrounding stock levels and harvest intensity across multiple user groups highlights the importance of a precautionary approach to management. This need and longfin biology suggest that area closure and territorial harvest rights are required for the effective protection of spawning biomass and the maintenance of exploitation.

A background to the New Zealand longfin eel fishery is provided in Section 2, followed by a description of the model and data sources in Section 3. Section 4 reports model results and sensitivity analysis. In addition, their implications for fishery management are discussed. A summary and concluding remarks are provided in Section 5.

2. New Zealand longfin eel fishery

The longfin eel was abundant in New Zealand freshwaters before heavy commercial harvesting began in the early 1970s. The shortfin and longfin comprised approximately 90 per cent of the total weight of freshwater fish caught in the country before this time (McDowall 1990). The longfin also remains the most widely distributed of any New Zealand freshwater fish species. Both shortfins and longfins may be found together in some habitats, such as coastal streams. However, inland eel populations often consist solely of longfins.

The biology of the longfin eel is particularly complex. Like all anguillids, the longfin is catadromous; that is, it spends a significant portion of its life in freshwater before reproducing in the sea. This makes the identification of a stock–recruitment relationship difficult. It also introduces a delay between spawning and the recruitment of young. Longfin eels are also semelparous (they produce all of their offspring at one time). This complicates fishery management as harvested longfins have yet to spawn. These difficulties are exacerbated through the extensive period (up to 105 years in some environments) that female longfins require to reach sexual maturity (Todd 1980; Jellyman 1995). These factors mean that the cumulative impact of even low levels of annual exploitation can have extremely deleterious effects on longfin populations (Hoyle and Jellyman 2002).

Most freshwater eels in New Zealand are caught using fyke nets that are fixed along the margins of waterways and left overnight in accordance with the nocturnal nature of eels. Periodic fishing can severely exploit an area because adult longfins inhabit a territory of less than 150 metres of a waterway's length (Chisnall and Kalish 1993). Also, eels may expand their home ranges into those of adjoining populations of low density; for example, those that have been heavily fished (Chisnall *et al.* 2003). Exploitation of one location may therefore impact on adjacent, though perhaps more inaccessible, populations. High territoriality also complicates stock assessment.

A minimum weight restriction of 220 g (enforced through minimum mesh sizes and escape tubes) was established in 1993 in an attempt to increase yield-per-recruit; however, yield-per-recruit for female longfins is maximised at a minimum weight limit at least twice this size (Hoyle and Jellyman 2002). In addition, this restriction does little to protect spawning biomass given that most female longfins will not reach sexual maturity until 20–50 years after attaining this weight. A maximum size limit of 4 kg was introduced into the South Island fishery in 1995 to protect longfins moving downstream before oceanic migration. (Maximum weight limits are enforced by fishers releasing larger eels following harvest.) Lower maximum weight restrictions of 1.5 kg (Chisnall and Hicks 1993) and 2 kg (Hoyle and Jellyman 2002) have been recommended, but these do not consider the opportunity cost associated with the higher price received for larger eels. In addition, longfin females will frequently be vulnerable to fishing gear for over 20 years before reaching the lower of these weights.

The South Island fishery entered the New Zealand quota management system (QMS) (Batstone and Sharp 1999; Newell *et al.* 2005) in 2000, and the North Island followed in 2003. This followed concern over the sustainability of harvests given the inadequate protection offered by existing limited entry policies. The allocation of property rights among customary, recreational and commercial fishers theoretically provides an incentive for conservation and efficient usage. However, a lack of biological data has led to the calculation of the TAC using the ‘average level of total landings over the longest recent period when the fishery has been relatively stable’ (New Zealand Ministry of Fisheries 2000, p. 8). The *total* harvest of freshwater eels in New Zealand has been relatively stable over the last two decades. This suggests that populations are being sustainably exploited. No meaningful species composition data has been obtained since 1992 (Jellyman *et al.* 2000), however, and the longfin is more likely to become overfished relative to the shortfin because of its slower maturation. The significant period required for young longfins to grow to commercial size (5–30 years) and the recruitment of young shortfins from spawning stocks outside of New Zealand (e.g., Australia) suggest that the recruitment failure of longfins would be difficult to detect in the short term.

3. The model

3.1 Model description

This section presents a bioeconomic model for the analysis of alternative management policies for the longfin fishery. The longfin stock of the lower Waikato River is used as a basis for this investigation because it is well studied in relation to many areas. It is also large enough to validate the use of a spawner–recruitment function, and it is representative of the entire fishery in that its longfin population is in an exploited state. This population is studied in aggregate form because of a lack of information, particularly in relation to individual variation. The members of each age class in the model therefore have uniform size, growth, mortality and patterns of sexual maturity. Eels in each cohort are described by a mean weight and length using data from Beentjes *et al.* (1997) and Chisnall *et al.* (2003). Therefore, the importance of size in the determination of important relationships is retained, even though eels are classified by age.

The assumed life cycle of longfins within the stock described by the model is shown in Figure 1. Sexually mature females take 6 months to swim from their freshwater home range to spawning grounds in the Pacific Ocean. The return of young is understood to take 18 months. It is therefore assumed that the elver stock entering the population in a given year depends on the spawning stock of 2 years before. These estimates are in line with analyses of the migratory patterns of longfins (e.g., Jellyman 1987).

Using the growth relationships mentioned above, eels are assumed to reach the current minimum harvestable weight at 11 years of age. Although this may be varied to investigate the effects of different weight restrictions, the division between *juvenile* and *adult* classes is always assumed to exist between eels of 0 and 10 years of age and eels of 11 years and above, as depicted in Figure 1. This division arises from dimorphism in dietary preference as eels below 40 cm in length (juveniles) feed from the substrate of their resident waterway, primarily eating insect larvae and molluscs, while larger longfins (adults) are predominantly piscivorous (fish eating) (Jellyman 1997). Adult growth within this model is classified as density-dependent since competition has a strong effect on the growth rates of larger eels (Chisnall and Hicks 1993; Jellyman 1997). In contrast, juvenile growth rates are density-independent.

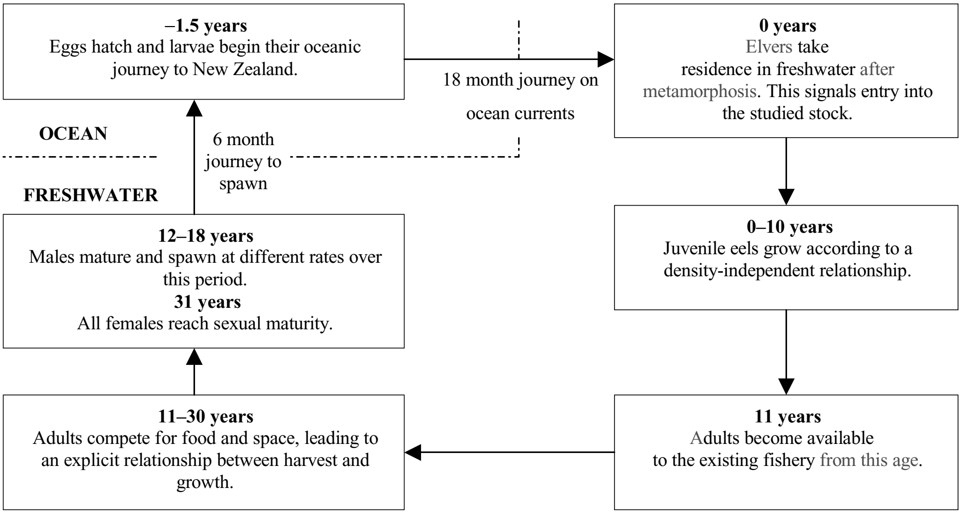

Figure 1 Generalised life cycle of the lower Waikato River eel stock.

Sexual dimorphism is also included to cater for differences in age at sexual maturity. The spawner–recruitment relationship in this model concerns only mature females, as they are considered the limiting factor for longfin recruitment (Jellyman *et al.* 2000; Hoyle and Jellyman 2002). In contrast, sexual maturity of males (which occurs at much younger ages) is represented through the exit of these eels in a way that is synonymous with natural mortality. All females are assumed to reach sexual maturity at 31 years of age. This is consistent with the mean size of female eels at maturation (1156 mm) (Todd 1980). In contrast, males reach sexual maturity between 12 and 18 years of age. An equal sex ratio for recruiting elvers is also assumed (Francis and Jellyman 1999; Jellyman *et al.* 2000).

The longfin stock is segregated according to the year-class descriptions in Figure 1. Individual cohorts are denoted by the subscript $k = [0, 1, 2, \ldots, k^s]$, where $k = 0$ represents the elver class and k^s denotes the eldest age group. For the youngest adult year class, $k^a = 11$. This cohort is consistent with the existing minimum weight restriction (220 g) and is consequently the youngest age class vulnerable to gear (denoted k^h). This assumption follows recommendations for higher minimum weight restrictions (Chisnall *et al.* 2003). Males reach sexual maturity and leave the system over ages $k^m = [12, 13, \ldots, 18]$. The eldest adult year class in the stock is $k^{s-1} = 30$. This is also the eldest harvestable cohort, denoted k^H. Those year classes defining the minimum (k^h) and maximum (k^H) size restrictions may be manipulated to reflect different management strategies.

The spawning cohort, which is not harvested (Hoyle and Jellyman 2002), is denoted $k^s = 31$. The delay between the escapement of spawning females and the return of elvers into the system is 2 years ($\omega = 2$). It is assumed that all eels between the ages of k^h and k^H have a constant rate of catchability and that perfect selectivity exists. The latter allows the comparison of prices that vary by age. This assumption is also appropriate given low handling mortality and the ability to sort catches at harvest.

It is assumed that a sole manager who aims to maximise net present value (NPV) owns the eel resource. This approach allows the consideration of important economic factors that are missing in biological models of the longfin fishery (see Jellyman *et al.* 2000; Hoyle and Jellyman 2002).

The estimation of a standard production function is problematic given the lack of harvest, stock and effort information of sufficient quality (Jellyman 1993). Anecdotal evidence suggests the presence of a stock-related cost externality and increasing marginal harvest costs. The latter are associated with localised depletion after harvest, higher gear-maintenance costs and difficulties in dealing with gear saturation as the catch increases. These factors may be incorporated through the inclusion of a harvest-stock ratio term in the cost function:

$$C_{k,t} = c\,Y_{k,t}\left(\frac{Y_{k,t}}{X_{k,t}}\right). \tag{1}$$

Here $C_{k,t}$ is the cost of harvesting cohort k at time t, c is the cost of harvest per tonne of eels, and $Y_{k,t}$ and $X_{k,t}$ denote the harvest and stock of cohort k at time t, respectively. The variables $Y_{k,t}$ and $X_{k,t}$ are both measured in tonnes. Harvest ($Y_{k,t}$) is only defined across those year classes exposed to gear; that is, cohorts across the interval $k = [k^h, \ldots, k^H]$.

This problem is cast as an infinite-horizon problem because there is no obvious terminal time. The present value of net benefits (π) accruing to the harvest of the studied stock is consequently:

$$\pi = \sum_{t=0}^{\infty}(1+r)^{-t}\left(\sum_{k=k^h}^{k^H} Y_{k,t}\left(P_k - c\left(\frac{Y_{k,t}}{X_{k,t}}\right)\right)\right). \tag{2}$$

Here r is the discount rate, and P_k is the price for harvested cohort k. Although, in reality, the price increases with eel size, here it is represented through a price that increases with age. This has a limited adverse effect on the model's output given the strong correlation between size and age (Chisnall *et al.* 2003).

The discounted net revenue function presented in Equation (2) is maximised subject to constraints representing changes in the eel stock over time. These occur through both natural effects (i.e., growth, sexual maturity, recruitment and natural mortality) and harvest. Following the life cycle depicted in Figure 1, these processes are represented in the following ways.

The weight of elvers reaching freshwater in a given year is dependent on the spawning biomass from ω years before. A Beverton–Holt spawner–recruitment relationship (Beverton and Holt 1957) seems the most appropriate because elvers compete for food and are highly susceptible to predation. The lag between escapement and the return of elvers into the system (ω) defines a delay-difference equation:

$$X_{0,t+\omega} = \frac{a\,X_{k^s,t}}{1 + b\,X_{k^s,t}}. \tag{3}$$

Here a and b are parameters describing the spawner–recruitment function. The corresponding recruitment constraint describes the size of the first adult cohort (k^a) as a function of the spawning biomass $\omega + k^a$ years before:

$$X_{k^a,t+\omega+k^a} = \frac{a X_{k^s,t}}{1 + b X_{k^s,t}} e^{\sum_{k=0}^{k^a} (G_k - M)}. \tag{4}$$

Here $\sum_{k=0}^{k^a} (G_k - M)$ is the total change in the size of a year class occurring through growth (G_k) and natural mortality (M) over the juvenile life stage.

The second constraint defines the growth dynamics of adult cohorts:

$$X_{k+1,t+\omega+k^a+1} = X_{k,t} e^{\phi(X_V - X_{AD}) - \mu(k^m) - M} - Y_{k,t}. \tag{5}$$

The size of adult cohorts may change through physical growth ($\phi(X_V - X_{AD})$), the loss of males at sexual maturity ($\mu(k^m)$), natural mortality (M) and harvest ($Y_{k,t}$). The rate of physical growth, $\phi(X_V - X_{AD})$, varies with population density. There is no growth at carrying capacity, which is assumed to be the level of unfished (virgin) biomass (X_V). However, as the current adult stock level ($X_{AD} = \sum_{k=k^a}^{k^s-1} X_{k,t}$) falls, growth will increase as eels compete less for food and space (De Leo and Gatto 1996; Jellyman 1997).

The feasibility constraints; $X_{k,t} \geq 0$, $Y_{k,t} \geq 0$, $X_V \geq X_{AD}$ and $X_{k,t} \geq Y_{k,t} \geq 0$; and Equations (2), (4) and (5) define a nontrivial discrete-time optimal control problem. This is solved with non-linear programming (NLP) using the CONOPT2 solver in the general algebraic modelling system (GAMS) (Brooke *et al.* 1998). This infinite-horizon problem is truncated to a finite horizon of significant length (65 years) to permit solution (Rowse 1995). Divergences from the steady-state near the end of the horizon are ignored as artefacts of truncation because the identification of a suitable terminal-value function is problematic. Experiments with the model with alternative initial conditions and extended planning horizons confirm the optimality of the equilibrium.

3.2 Estimation of parameters for base model

The estimation of exogenous parameters is briefly described here. However, a more detailed description is available at www.are.uwa.edu.au/home/eel_model. Parameter values for the base model are presented in Table 1.

The real discount rate (r) is 5 per cent: the mean, long-term New Zealand government bond rate for December 1994 to December 2001. The five major eel processors in New Zealand were interviewed to obtain price information (P_k). Harvesters licensed to fish in the lower Waikato River were surveyed to identify the cost of harvesting (c).

Geographical information systems were used to identify the total area of the study region. An estimate of the virgin stock level ($X_V = 690$ tonnes) was then obtained using this and density data from Chisnall *et al.* (2003). A simulation model was developed using this estimate of the virgin stock and was used to generate equilibrium values of the adult population (X_{AD}) and corresponding growth rates ($g(X_{AD})$) for a wide range of fishing mortality values (see Campbell *et al.* 1993). Data series for ($X_V - X_{AD}$) and $g(X_{AD})$ were then regressed to identify a value for the density-dependent growth

Table 1 Description of exogenous parameters for the base model

Parameter	Description	Value
r	Discount rate	$r = 0.05$
P_k	Price for harvested cohort k	$P_k = \$4350/\text{tonne}$ for $k = 11–14$, $P_k = \$4540/\text{tonne}$ for $k = 15–18$, $P_k = \$4850/\text{tonne}$ for $k = 19–20$, $P_k = \$5950/\text{tonne}$ for $k = 21–30$
c	Harvest cost/tonne	$c = \$2071/\text{tonne}$
ϕ	Density-dependent growth factor	$\phi = 0.0004$
G_k	Annual rate of growth for juvenile cohort k	Estimation of these parameters is described at: www.are.uwa.edu.au/home/eel_model
M	Annual rate of natural mortality	$M = 0.04$
$\sum_{k=0}^{k^a} (G_k - M)$	Survival rate for juveniles	$\sum_{k=0}^{k^a} (G_k - M) = 4.65$
$\mu(k^m)$	Annual rate of male migration as a function of age	$\mu(12) = 0.015$, $\mu(13) = 0.035$, $\mu(14) = 0.105$, $\mu(15) = 0.114$, $\mu(16) = 0.177$, $\mu(17) = 0.057$, $\mu(18) = 0.025$
a	Spawner–recruitment function parameter	$a = 0.014$
b	Spawner–recruitment function parameter	$b = 0.067$

factor (ϕ). Initial cohort sizes were identified using this model and an estimate of the intensity of past exploitation. A rate of 20 per cent was identified by Jellyman *et al.* (2000).

Growth rates for juveniles were identified using data from Chisnall *et al.* (2003). The rate of natural mortality (for both juveniles and adults) was taken from Francis and Jellyman (1999). The instantaneous rate of male migration for each affected age class ($\mu(k^m)$) was calculated from data in Todd (1980) and Chisnall *et al.* (2003). A lack of data prevented the statistical estimation of a suitable spawner–recruitment function. An equilibrium method derived by Ricker (1975) was consequently extended and applied to the studied stock through substitution of the population dynamics described above. This allowed identification of the parameters a and b in Equation (4).

4. Results and discussion

This section presents results from the base model and sensitivity analysis. It also discusses their implications for the management of the longfin fishery.

4.1 Base model output

The studied fishery recovers reasonably slowly from its depleted state under optimal management (Figure 2a). The equilibrium stock level ($X^* = 183$ tonnes) is much lower than the virgin biomass ($X_V = 690$ tonnes). This is consistent with surplus production, and consequently steady-state harvest, directly benefiting from the stimulation of growth rates at low population density. In contrast, recruitment plays a lesser role as annual growth has a cumulative effect on cohort biomass over time, due to the

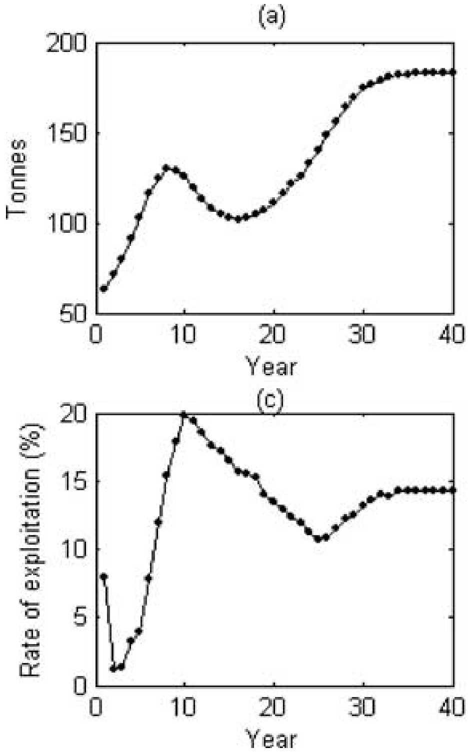

Figure 2 Optimal trajectories from the base run of the model for (a) stock, (b) harvest and (c) the rate of exploitation.

longevity of longfins; therefore, a significant adult population may establish even when elver return is low. This is demonstrated by the fact that equilibrium annual recruitment is just over 36 kg.

The optimal recovery policy requires that harvesting be maintained at low levels for the first 5 years (Figure 2b). This permits spawning biomass to increase to its equilibrium value (3 tonnes) in just 6 years. Spawning cohorts are maintained at low levels given the lesser importance of recruitment for surplus production outlined above and since investment in these year classes will directly reduce harvest because of semelparity. In contrast to the rapid adjustment of spawning biomass, it takes a significant amount of time for the stock levels of the other age classes to reach equilibrium (33 years). This follows the delay between spawning and the entry of corresponding young into the fishery ($k^a + \omega = 13$) and the time required for the initial exploited cohorts to move through the population ($k^s - k^a = 20$). This demonstrates the significant period required for the recovery of a semelparous, long-lived fish population.

This recovery is heavily dependent on effective restriction of the exploitation rate (Figure 2c). The limitations of the current quota-setting methodology are apparent given that an estimate of historical exploitation (20 per cent) (Jellyman *et al.* 2000) can only be supported in a single year (year 10). Population uncertainty, the poor quality of catch data across all user groups (harvest uncertainty), and potential overfishing of quota because of poor enforcement complicate ITQ management. Enforcement of different rates of exploitation in the model identifies that NPV decreases by over half and the stock declines to single figures within 10 years if sustained exploitation rates greater than 10 per cent are permitted prior to recovery. This demonstrates the sensitivity of longfin populations and the profitability of their fisheries to the miscalculation of sustainable catch. Density-dependent growth factors and harvest costs help to prevent extinction in the model. Their validation and measurement are consequently critical in the provision of greater insight into the long-term effects of current methods of quota estimation.

Optimal equilibrium management incorporates a significant harvest rate, a low equilibrium stock and a small spawning cohort. These are intuitive given the assumed population dynamics. The effects of negative shocks, such as recruitment failures, have not been considered though and could be expected to moderate these recommended strategies, perhaps significantly. Stochastic features are not incorporated here because of poor information and the robustness of the model to significant changes in a number of important parameters (see Section 4.2). However, a critical area for further work is identifying the implications of uncertainty for the optimal harvest profile, particularly if the quantity and quality of recruitment data improves.

The optimal equilibrium exploitation rate in the base model is 14.7 per cent. This may be compared to the long-run exploitation rate that maximises maximum sustainable yield (MSY) through enforcing a zero discount rate ($r = 0$) in the model. The equilibrium stock ($X_{MSY} = 244$ tonnes) and harvest ($Y_{MSY} = 15.6$ tonnes) levels are significantly higher (33 per cent and 11 per cent respectively) after such a modification. The greater relative change in the stock level, following the removal of discounting, leads to a dramatic decrease in the optimal exploitation rate to 6.4 per cent. This result is comparable to estimates (5–8 per cent) from biological models that do not incorporate economic and spawner–recruitment relationships (Jellyman *et al.* 2000; Hoyle and Jellyman 2002).

Under optimal management in the base case, harvesting is restricted to the most valuable age classes: cohorts 21–30, which are all female (see Table 1 for relative monetary values). No harvest of less valuable year classes (220 g to 1.4 kg, or ages 11 to 20 within this model) indicates that growth overfishing may be overcome through increasing the minimum weight restriction to the lower bound of those age groups that receive the highest price (1.5 kg). This demonstrates the inefficiency of current exploitation as the majority of harvested eels now weigh less than 500 g (Beentjes 1999). Such a high minimum weight restriction would protect males from exploitation as they all spawn beneath this weight (Todd 1980). However, the harvest of females would have to be limited to moderate levels if spawning biomass is to be maintained. Moreover, this minimum would restrict harvest significantly in multiple-species eel fisheries because shortfins are also semelparous, 95 per cent spawn beneath this weight (Jellyman 1993), and gear cannot select between species. Nonetheless, this finding suggests that a higher minimum weight restriction will improve efficiency in the longfin fishery.

Chisnall and Hicks (1993), Jellyman *et al.* (2000) and Hoyle and Jellyman (2002) reported that maximum weight limits of 2 kg or lower were required to significantly increase spawning biomass. Restriction of harvest to the most valuable cohorts suggests,

Table 2 Net present value, equilibrium harvest and spawning biomass for alternative maximum weight restrictions

Maximum weight restriction	Net present value ($)	Equilibrium harvest (tonnes)	Spawning biomass (tonnes)
2 kg	487 100	11.2	8
3 kg	684 500	12.6	6
4 kg	796 600	13.9	4
No restriction	877 600	14	3

though, that lower maximum weight restrictions may significantly decrease the value of the resource through constraining the harvest of the most valuable age classes. In line with this supposition, the model's output demonstrates that NPV increases significantly as maximum weight limits bind less strongly (Table 2). Spawning biomass does fall as maximum weight limits increase given the trade-off between the harvest and spawning value of a semelparous individual (Table 2). However, recruitment increases are not sufficient to surpass the benefits accruing to the vulnerability of a high number of year classes. This reinforces the importance of physical growth to exploitation in this fishery. These findings indicate that the 4 kg limit utilised in the South Island fishery may be preferable to a lower restriction from an economic standpoint, a proposition directly contrary to recommendations from fish biologists.

4.2 Sensitivity analysis

It is important to explore the sensitivity of output to changes in key relationships and parameters within the model because of the uncertainty involved in their estimation and specification.

4.2.1 Spawner–recruitment function parameters

The importance of physical growth to equilibrium exploitation is reinforced through the low sensitivity of the profitability of the fishery to significant variation in the parameters defining the spawner–recruitment function (a and b). NPV is altered by a maximum of 5 per cent with a 20 per cent change in the magnitude of these estimates. This robustness follows from the multiple-cohort nature of longfin populations, as the harvest of annual growth over a significant number of cohorts (year classes 21–30) permits a reasonable level of exploitation despite the low recruitment of young.

4.2.2 Price, cost and discount rate parameters

Recommendations for different weight restrictions are based on a harvest profile that restricts harvesting to older, more valuable year classes. The optimality of harvesting older cohorts may arise from the greater time they are permitted to grow, contribute to the lowering of harvesting costs before exploitation and the price premium they receive. The majority of harvesting remains concentrated on these older cohorts, though,

Table 3 Changes in population, spawning biomass, rate of exploitation and NPV for alternative values of the density-dependent growth parameter (ϕ), relative to the base model scenario ($\phi = 0.0004$)

Density-dependent growth parameter	Steady-state stock (% change)	Steady-state spawning biomass (% change)	Steady-state rate of harvest (% change)	NPV (% change)
$\phi = 0.00036$ (10% decrease)	−10.9	N/C	−5.6	−17.8
$\phi = 0.00038$ (5% decrease)	−4.9	N/C	−3.5	−8.1
$\phi = 0.00042$ (5% increase)	+3.8	+33.3	+9.8	+12.5
$\phi = 0.00044$ (10% increase)	+7.7	+33.3	+11.5	+23.3

NPV, net present value and N/C, no change.

once a constant price is utilised across all year classes.[1] Also, a 25 per cent increase and decrease relative to the estimated cost and discount rate parameters yields little change.[2] (The latter is important because a higher discount rate reduces the value of an unharvested fish at time t.) These results imply that the optimal age structure is primarily determined through biological rather than economic factors. The optimal harvest rate is also quite robust to these changes. These factors reinforce the suitability of the recommended strategy.

4.2.3 Rate of density-dependent growth

In contrast, the base model is sensitive to changes in the density-dependent growth parameter, with large deviations observable for harvest and consequently NPV (Table 3). This reinforces the importance of physical growth, as the bioeconomic principle in which slower or faster growth allows lower or higher equilibrium yield is demonstrated. The primary role of physical growth relative to recruitment is also apparent as, in contrast to this result, the model is robust to significant changes in the parameters defining the spawner–recruitment relationship.

4.2.4 Incorporation of density-independent growth rates

It is interesting to test the sensitivity of output to constant adult growth rates, particularly because existing models of the longfin fishery incorporate such a relationship. Fixed growth rates are calculated utilising information from Chisnall *et al.* (2003). Stock recovery is significantly slower relative to the base model (Figure 3a), as expected. In addition, exploitation must periodically be maintained at low levels in order for this recovery to occur (Figure 3b). However, no equilibrium incorporating a constant harvest rate is ever experienced. The resultant harvest profile has a large effect on total catch, which halves over the first 50 years. NPV consequently decreases by

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[1] The mean price from the data in Table 1 (P_k = \$NZ4923) is used across all year classes to identify this result.

[2] The cost parameters utilised in the sensitivity analysis are \$NZ1553 and \$NZ2589. The discount rate parameters used are 3.75 and 6.25 per cent. Wide deviations in the discount rate are not expected given a policy commitment to low and stable inflation in the *Reserve Bank of New Zealand Act (1989)*.

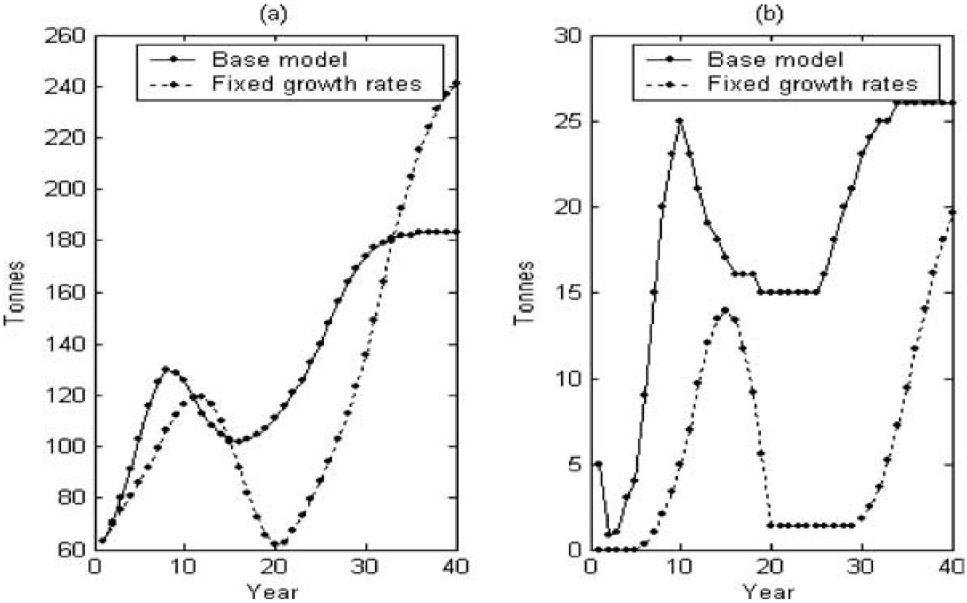

Figure 3 Optimal trajectories for (a) stock and (b) harvest for density-dependent and fixed growth rates.

approximately 60 per cent. These results reinforce the importance of density-dependent growth in the provision of surplus production for annual harvests within this model.

4.3 Management implications

Significant concern over the depletion of longfin eel fisheries hastened their entry into the QMS. ITQ systems have definite benefits given property-right allocation, the experience of New Zealand fishery managers in the design and implementation of these systems and the process of structural adjustment aided through quota trading (Newell *et al.* 2005). Their successful application to the longfin fishery is problematic for several reasons though:

1. Competition remains among commercial harvesters as larger eels are sought to maximise quota value (high-grading). Recreational and customary harvesters also compete with commercial fishers, given their preference for larger longfins (Jellyman 1993).
2. Surplus production may be too low to sustain a profitable fishery if growth is independent of stock density.
3. Current stock assessment procedures are insufficient to calculate sustainable catch.
4. Results suggest that the use of past harvest data to formulate TACs would not permit stock recovery and may lead to stock collapse. Such a collapse is particularly likely if enforcement is poor and growth and harvest costs do not sufficiently promote population stability.

An attractive alternative to an ITQ system is the zoning of areas within the fishery. Area closures are particularly attractive because of the difficulties encountered in the application of weight restrictions, significant larval spillover (Chisnall and Hicks 1993; Jellyman *et al.* 2000; Hoyle and Jellyman 2002), territoriality and the potential to protect against severe overexploitation arising from population and harvest uncertainty (Lauck *et al.* 1998; Gerber *et al.* 2003). The latter is critical given the vulnerability of females to gear across multiple years and competition among harvesters for larger eels in fished areas.

It is difficult to identify the required size of unfished areas because of a lack of information. Existing protected areas are constrained by barriers to migration (e.g., hydroelectric development) and low productivity (Jellyman 1993). For example, 83 per cent of protected lakes and lagoons in New Zealand are presently inaccessible to elvers (Annala *et al.* 2001). These factors suggest that a substantial decrease in the fishery may be required. Decision theory is particularly useful in optimal reserve design (Possingham *et al.* 2000) but poor data limits its usefulness in this instance. Active adaptive management, whereby a set of closed areas is constantly refined in a non-linear fashion through enforcement, evaluation and updating, is a valuable alternative (Grafton and Kompas 2005).

The use of closed waterways, such as large river systems, has significant appeal for several reasons:

1. Spawners emanating from existing unfished areas are often required to pass through areas open to harvest during migration. The use of closed waterways would circumvent a need for seasonal maximum weight restrictions and the associated enforcement effort required to protect these eels.
2. Protected areas will not be restricted to inland areas where productivity is low (Jellyman 1993).
3. Diffusion of eels from closed areas into exploited populations (Chisnall *et al.* 2003) would be avoided.
4. Closed waterways would safeguard both genders given differences in gender composition between coastal and inland areas.

Area closures could be complemented with territorial use rights in fishing (TURF) that allocate exclusive harvest rights over an area to a user or user group (Christy 1982). The limited mobility of the longfin and the capacity to allocate traditional fishing grounds, or rohe, to customary fishers are obvious benefits. The latter concept complements the use of area closures, which were historically enforced by customary fishers (Best 1986). Use of area quotas conceptually provides a stronger form of property right than that implemented through an ITQ system. The territoriality, high catchability and longevity of longfins mean that there is a large incentive to overexploit an area if property rights are not spatially defined. Exclusivity may be particularly difficult to enforce though.

Management of fished areas may benefit directly from results identified in this paper. First, a higher minimum weight restriction will help to maximise the benefits received from growth. Second, removal of maximum weight restrictions will permit the harvest of the most valuable individuals. Maintenance of unfished areas will help to overcome any detrimental effect that the removal of a maximum weight restriction may have on spawning biomass, particularly if selectivity is poor. Last, recovery may be achieved through limiting total catch until populations can support constant exploitation. Short-term costs are likely to be borne, although their distribution and magnitude are difficult to estimate a priori.

5. Summary and conclusions

A number of studies of the New Zealand longfin eel fishery (Jellyman *et al.* 2000; Hoyle and Jellyman 2002) have been implemented amid significant concern over the sustainability of this and other anguillid fisheries throughout the world. However, this paper is the first to address important economic and dynamic effects, particularly density-dependent growth and spawner–recruitment relationships.

Longfin biology and significant uncertainty highlight the importance of area closures for the precautionary management of longfin populations. This is necessary because the use of past harvest levels to calculate TACs will place longfin populations at increased risk of collapse. The allocation of territorial user rights would be highly advantageous. In these fished areas, commercial fishers may benefit appreciably from restricting harvest to older eels.

Significant levels of harvest may be supported if longfin fisheries are permitted to recover. However, this proposition depends on the true strength of the interaction between population density and longfin growth, which is presently unknown. Moreover, although the optimal harvest rate is robust to significant changes in a number of important economic and biological parameters, its sensitivity to environmental shocks and/or harvest uncertainty has not been tested. Both are critical areas for further work.

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