THE SUSTAINABLE MANAGEMENT OF THE NEW ZEALAND LONGFIN EEL: A BIOECONOMIC ANALYSIS

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ABSTRACT

Annual recruitment of the New Zealand longfin eel (*Anguilla dieffenbachii*) has declined by around 75 percent since heavy levels of commercial fishing began in the early 1970s. Longfin eels live in freshwater for many years, sometimes over one hundred, before reaching sexual maturity and migrating to oceanic spawning grounds. Longfin eels are semelparous, in that they die after making only a single reproductive contribution following migration. Late maturation and semelparity render longfin populations extremely sensitive to recruitment overfishing. Consequently, poorly defined property rights and fragmented regulation have permitted multiple user groups, but primarily the commercial fishery, to reduce these stocks to the point of near-collapse.

In this research, a deterministic multiple-cohort bioeconomic model is developed and applied to a longfin eel population to investigate sustainable management strategies for the fishery, subject to its biological and economic characteristics. The optimisation framework incorporates density-dependent growth and spawner-recruitment relationships and a delay-difference equation to express the significant lag between the sexual maturity of adults and the vulnerability of corresponding young to the fishery. The model also permits the investigation of alternative weight restrictions and a price that varies with age/size.

The model demonstrates the insufficiency of using past harvests to calculate sustainable catch, as done recently for the South Island fishery. The model results also indicate the need for a minimum weight restriction higher than that maintained under the existing regulatory system. The importance of no maximum weight restriction is also identified. Additionally, the model results indicate that there is a significant inverse relationship between the level of exploitation and the annual breeding population, since no harvested eel has ever spawned. The sensitivity of longfin eel populations to recruitment overfishing is greater in reality due to uncertainty, competition among harvesters, price and harvest incentives, and this specie's biology. These factors suggest that the use of any harvest-based regulatory system without significant investment in area closure will fail to protect longfin eel stocks through the recovery and maintenance of spawning biomass.

The analysis identifies the need for an integrated management strategy, incorporating area closures, for rebuilding and maintaining spawning biomass, and the use of ITQ management in open fisheries to aid the allocation of fishing rights among users. Efficient management of these open areas requires a higher minimum weight limit than under the current management system, and no maximum weight restriction. The calculation of sustainable harvest levels remains problematic due to poor information; however, active adaptive management may be used to work towards their identification. This approach might be aided by density-dependent growth, which would assist the recovery of populations if sustainable harvest were overestimated. Additionally, spawners from closed populations would help to safeguard against recruitment overfishing during the investigation of sustainable exploitation rates. This integrated policy represents a biologically sound and economically relevant management strategy that has the potential to sustain longfin populations and their harvest indefinitely.

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1. INTRODUCTION

1.1 Background

Freshwater eel populations throughout the Northern Hemisphere have experienced significant declines over the last twenty to thirty years (Dutil et al. 1989; Moriarty 1990; Dekker 2000), with large decreases observable in species from North America (*Anguilla rostrata*), Europe (*A. anguilla*), and Asia (*A. japonica*) (Queensland Department of Primary Industries 2002). It is difficult to identify a single cause for these declines (Moriarty and Dekker 1997), given the combined deleterious effects of commercial harvest, habitat loss, physical barriers to adult migration (such as hydroelectric dams), a changing ocean environment, and pollution (Moriarty 1996). But the severe impact that frequent exploitation has on freshwater eel populations is illustrated through the rapid decline of a Southern Hemisphere stock, the New Zealand longfin eel (*A. dieffenbachii*). Biologists estimate that recruitment of this species has decreased by around 75 percent since the commencement of heavy commercial harvest, stating "… heavy fishing pressure since 1970 has reduced the stocks of large female longfins and … this is responsible for the decline in glass eel recruitment" (Jellyman et al. 2000, p41).

The longfin is a catadromous species, with individuals spending a significant proportion of their lives in freshwater before migrating to oceanic spawning grounds at sexual maturity. These animals die after making a single reproductive contribution (semelparity), while their young return to New Zealand on ocean currents as larvae. Following metamorphosis in estuaries, these young inhabit limited home ranges in freshwater until attaining sexual maturity (Burnet 1969; Chisnall and Kalish 1993), which may take females up to 106 years (Todd 1980; Jellyman 1995).

This reproductive cycle renders longfins particularly susceptible to overfishing. Semelparity complicates the management of the fishery, as longfins who succumb to fishing mortality have never contributed to spawning (McCleave 2001; Hoyle and Jellyman 2002). This difficulty is exacerbated through the extensive period that these animals require to reach sexual maturity due to slow growth, as individual eels may have to avoid annual harvest for up to eighty years before spawning. Slow growth also means that longfins become vulnerable to the commercial fishery up to thirty years after entry into freshwater (Annala et al. 2001), therefore significant yields may be supported while the entry of young into the fishery is declining (Hoyle and Jellyman 2002). This renders the longfin fishery particularly prone to sudden collapse. Stock assessment is also difficult given the wide distribution of longfins, heterogeneous growth between environments, and the territoriality of this species, which renders mark-recapture methods (Ricker 1975) ineffective (Annala et al. 2001).

Other factors put pressure on longfin populations. Poorly defined property rights have promoted competition for eels among harvesters belonging to recreational, cultural, and commercial user groups. Consistent global demand for eel products has supported relatively high and stable prices (McDowall 1990; Jellyman 1993) and stimulated overexploitation, given that fixed and variable harvesting costs are comparatively low. The prices paid to harvesters by processors are also generally increasing in the size of individual eels, providing a market incentive for fishers to target those animals that are closer to sexual maturity. This promotes the fishing of unexploited waterways where these larger animals are more common and catchability is high (McDowall 1990; Beentjes 1999; Jellyman et al. 2000). The nature of gear also promotes the exploitation of longfins, since the fyke nets that are primarily used within this fishery are efficient and well suited to placement in the preferred habitat of larger eels.

These factors have led to a substantial decline in longfin populations, impacting on society in a number of ways. First, this decline threatens the future sustainability of a considerable proportion of catch (35 to 40 percent)[1] in a small, but significant, commercial fishery. Second, the decline of longfin stocks represents a significant risk to the future maintenance of biodiversity. New Zealand has an obligation to conserve such endemic species under the Convention on Biological Diversity (CBD), which it ratified in 1993 (Department of Conservation 2002). Third, longfins are ecologically valuable as the top predators within their freshwater ecosystems. Their removal would consequently lead to ecological instability in many of these habitats. Fourth, the longfin

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[1] The remainder of harvest consists of the earlier maturing shortfin (*A. australis*) and the seldomly encountered Australian longfin (*A. reinhardtii*) eel.

supports a significant cultural fishery. The Ministry of Fisheries has a commitment to protect such fisheries, given their obligations under the 1840 Treaty of Waitangi (Te Waka a Maui me ona Toka Mahi Tuna 1996). Last, longfins also support a small recreational fishery. It is apparent that longfin stocks provide many values aside from those accruing to commercial harvest. This analysis therefore focuses on the identification of sustainable management strategies for the recovery and maintenance of these overexploited populations to secure the provision of these values. This study focuses solely on the commercial fishery, as this is the principle cause of this specie's overfishing (Jellyman et al. 2000; Hoyle and Jellyman 2002).

1.2 Management policies

The determination of sustainable management strategies is necessary due to the inability of existing regulations to protect longfins from overharvest. National minimum weight restrictions, a maximum weight limit in the South Island fishery, various gear restrictions, and a moratorium on fishing permits have all been of some benefit. However, due to a number of inherent deficiencies, none of these policies has significantly limited harvest or increased spawning biomass (Jellyman et al. 2000; Chisnall et al. 2002; Hoyle and Jellyman 2002). A more efficient and effective regulatory system is therefore needed.

Many analysts support the use of the Quota Management System (QMS), New Zealand's existing Individual Transferable Quota (ITQ) scheme, to protect the longfin eel resource (Town 1986; Te Waka a Maui me ona Toka Mahi Tuna 1996; Annala et al. 2001). Customary, recreational, and commercial fishers are allocated proportions of a Total Allowable Catch (TAC), which is based upon biological estimates of sustainable harvest, within the QMS. This theoretically provides an incentive for conservation through the allocation of property rights and the self-interest of each fisher to manage the resource sustainably. The QMS provides a firm foundation on which to base future management, given its significant history and the experience of fishery managers in the design and implementation of these systems.

While this system has a number of apparent strengths, the application of the QMS to the freshwater eel fishery has thus far lacked coherency. While the South Island fishery was entered into the QMS on 1 October 2000 as part of a Treaty of Waitangi settlement, the North Island has yet to be placed under its control (although this is planned for some time in 2003). In addition, the majority of South Island TACs were established using the "average level of total landings over the longest recent period when the fishery has been relatively stable" (Ministry of Fisheries 2000, p8) due to a lack of biological information. This primarily involved the use of a mean harvest level calculated from the five previous fishing years (1994/95 to 1998/99). It is anticipated that a similar method will be adopted in the North Island.

Although the calculation of reasonably accurate TACs utilising other methods is problematic due to a scarcity of information, the stability of harvest portrayed by regional figures does not represent reality because of poor information. For example, no accurate data on species composition has been available since 1992 (Jellyman et al. 2000). A change in the measurement of the unit of effort in 1989 and high annual variability in regional series also complicate the use of catch and effort data to identify trends within the fishery (Jellyman 1993). Additionally, while harvest may remain stable as shortfin populations may be sustained from the recruitment of young arising from spawning stocks outside of New Zealand (for example from Australia) (Annala et al. 2001), longfins are endemic and thus recruitment from nations with more robust populations cannot occur. Recruitment failure is also difficult to detect in the short-term, given that yields may be sustained as young longfins take between five and thirty years to enter the fishery (Annala et al. 2001).

Additional factors complicate the effective implementation of ITQ management in this fishery. It is possible that the use of annual quotas is fundamentally flawed as a tool for the protection of the long-lived longfin, as surplus production in these populations may be too low to sustain both spawning biomass and a profitable fishery because of slow growth and semelparity. Furthermore, while ITQs theoretically secure incentives for conservation through the definition of property rights, it is anticipated that the "race-to-fish" will remain as fishers compete for larger, more valuable eels to maximise the value of quota (high-grading). This may put substantial pressure on spawning populations, especially in unexploited areas where catchability is high. All user groups may also compete for more accessible fisheries in order to lower travelling costs, although yields here will be low because of past harvest and the sensitivity of longfin populations to exploitation (Jellyman et al. 2000; Hoyle and Jellyman 2002). These factors suggest that some analysis of the use of ITQ management in this fishery is required.

Uncertainty and the sensitivity of longfins to exploitation has motivated a number of authors to suggest that area closures are the only means of providing enough spawners to maintain these populations (Chisnall and Hicks 1993; Jellyman 1993; Jellyman et al. 2000; Hoyle and Jellyman 2002). Yet the adoption of this policy has potentially significant ramifications for all user groups as there is a present deficiency of closed areas in which migration is unimpeded and both growth and populations are high enough to provide significant levels of individuals reaching sexual maturity (Jellyman 1993; Annala et al. 2001). Area closures consequently have the potential to decrease the size of the available fishery and the provision of many values accruing to its use. Additionally, a number of biological characteristics may work against their successful implementation (for example, the diffusion of eels into harvested areas) (Chisnall et al. 2002).

It is apparent from this discussion that further analysis of suitable regulatory strategies is required. This research therefore seeks to complement the biological studies of Jellyman et al. (2000) and Hoyle and Jellyman (2002) through the investigation of management policies for the longfin fishery from a bioeconomic perspective.

1.3 Objectives of this research

The primary objective of this research is to investigate sustainable management strategies for the New Zealand longfin eel fishery while considering both its biological and economic characteristics. This investigation will thereby provide policy-makers with valuable additional information for the development of regulatory policies for these overexploited populations.

This study seeks to achieve this primary objective through the satisfaction of a number of secondary objectives. These are:

1. identify the important biological and economic factors contributing to the overharvest of the New Zealand longfin eel

2. develop a multiple-cohort bioeconomic model to gain insight into sustainable harvest profiles and suitable regulatory policies for the fishery, and
3. recommend biologically sound and economically relevant management strategies based on these findings.

This research will be useful to policy-makers responsible for the regulation of the New Zealand freshwater eel fishery, as well as those responsible for anguillid management throughout the world. The development of the multiple-cohort bioeconomic model will also provide some insight into the long-term effects of current quota-setting methodologies for the existing eel fishery, together with the identification of management policies suitable for practical application. Additionally, the model will provide the basis for future exploration into the optimal management of long-lived multiple-cohort stocks, especially those of a semelparous nature.

1.4 Organisation of this research

The research presented within this thesis is organised as follows. Chapter 2 describes the biology of the longfin and how this contributes to its susceptibility to overexploitation. Chapter 3 provides an overview of the New Zealand commercial eel fishery and its management. Chapter 4 describes the development of the theoretical foundations of modelling the optimal exploitation of age-structured populations through an analysis of past and present literature. Chapter 5 presents the construction and partial solution of the analytical model used in this research. Chapter 6 outlines the numerical model and the sources of data. Chapter 7 provides the model results and a discussion of the findings. Finally, Chapter 8 provides a discussion of the critical elements of this work, examines the policy implications arising from this study, discusses various limitations pertaining to this analysis, and suggests areas in which further research is required.

2. BIOLOGY OF THE LONGFIN EEL

2.1 Introduction

The biology of the longfin eel is very similar to that of many anguillids throughout the world but it appears as a special case of this genus due to the extreme degree to which these biological processes occur. For example, the longfin is the longest-living species of freshwater eel, with the oldest recorded individual living to 106 years of age (Jellyman 1995).

In this chapter the biology of the longfin eel is reviewed, with particular emphasis placed on those factors that complicate its management. The general biology and distribution of the longfin is presented and then the reproductive biology of this animal is described. The high plasticity of longfin growth is then examined.

2.2 General biology

The longfin eel is an elongated fish, with continuous dorsal, caudal, and anal fins along its belly and back and around its tail (Figure 2.1) (McDowall 1990). The dorsal fin extends much further along the top of its body than the anal fin does on the bottom, in contrast to the shortfin. Adult eels are variable in colour, usually having a dark brown or black dorsal area and a yellowish-grey underside. The head and back of longfins becomes a dark grey-black colour and the belly becomes dusky grey-white at sexual maturity (McDowall 1990).

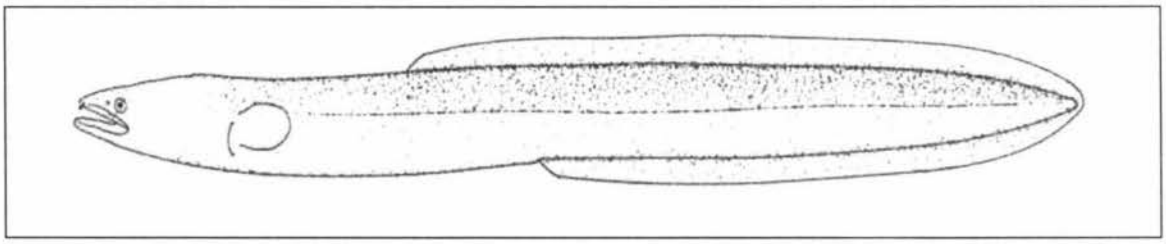

Figure 2.1. The body shape of a typical longfin eel (Paul 2000).

2.2.1 Distribution

The longfin is the most widely distributed of any New Zealand freshwater fish species (see Figure 2.2, where black dots represent reported incidence). It is estimated that before heavy commercial harvest began in the early 1970s, the shortfin and the longfin composed around 90 percent of the total weight of freshwater fish found here (McDowall 1990).

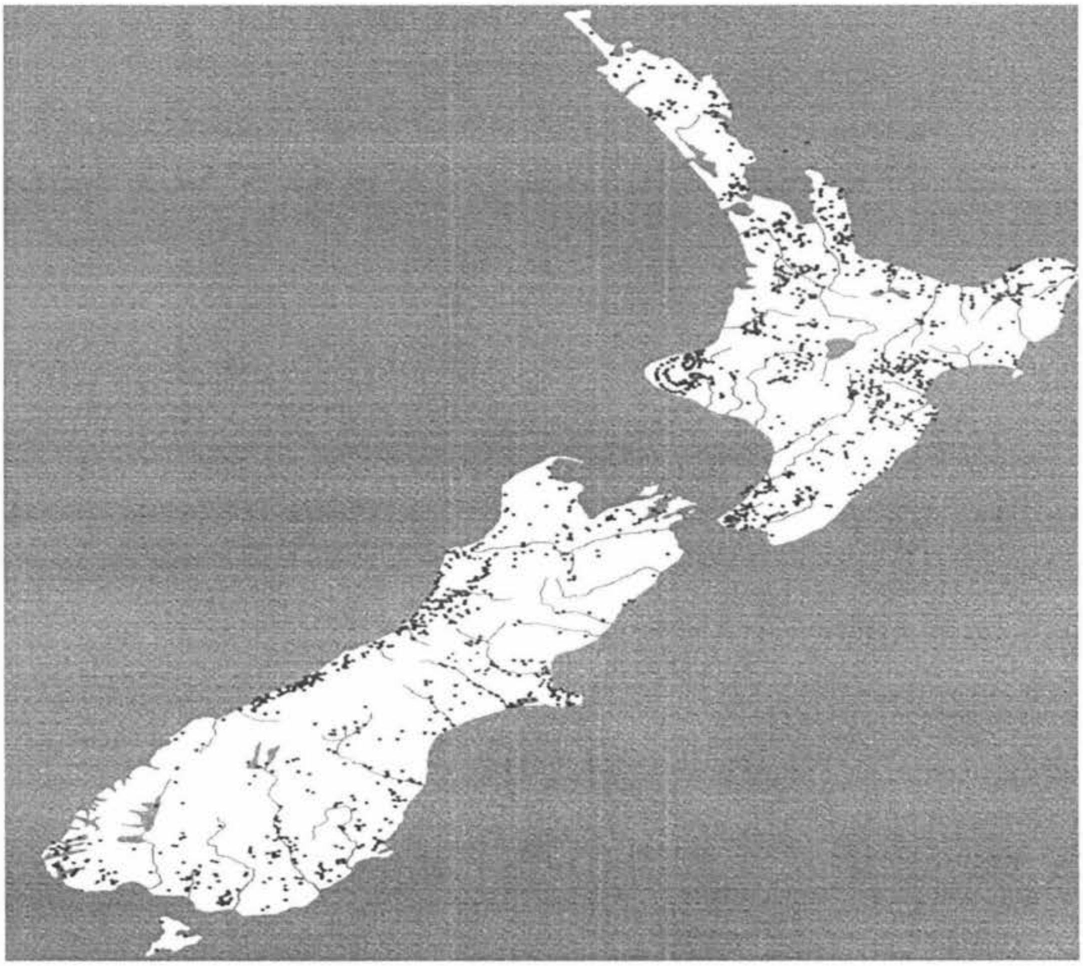

Figure 2.2. National distribution of the longfin eel (McDowall 1990).

The wide distribution and historical abundance of New Zealand's eels suggested in the past that these stocks might be able to support a significant level of exploitation. This attitude has promoted the decline of longfin populations, as user groups have failed to recognise their sensitivity to continued fishing.

Longfins inhabit locations from sea level up to any lakes and waterways they can reach as young elvers, this term denoting the freshwater phase of young after their journey on ocean currents as larvae and glass eels. Even though the longfin is found in many environments, including coastal lagoons, it usually prefers faster-flowing stony waterways and inland lakes. In comparison, shortfin eels are predominantly found in more stable, slower-flowing waterways in lowland areas and do not penetrate as far inland. Although both are often found together, populations in inland areas frequently consist solely of longfins.

Even though distinct habitat preferences exist today, it is believed that longfin eels were the dominant species in coastal areas prior to extensive habitat loss and commercial fishing (Chisnall 1994; Chisnall et al. 2002). This is also reflected in harvest histories, as over time the commercial eel fishery has moved from the exploitation of large female longfins to an increasing reliance on the capture of male shortfins (Chisnall 1994; Beentjes and Chisnall 1997). This demonstrates that anthropogenic influences have detrimentally affected eel populations to the extent that longfins are no longer the dominant species in New Zealand's freshwater environments.

2.2.2 Feeding

The diet of longfin eels has been described in a number of studies (Burnet 1968; Jellyman 1989, 1996). Jellyman (1989) identified that the diet of longfins is broad and flexible. Smaller specimens (less than 45 cm in length) obtain food from the substrate of their resident waterway, primarily feeding on insect larvae, worms, and amphipods. Alternatively, larger eels are predominantly piscivorous, with juvenile brown trout forming a substantial part of their diet (Jellyman 1996).

The morphology of large longfin eels is suited to predation. Their acute sense of smell, ability to tear meat from carcasses through rotational spinning (Jellyman 1996), extensile stomach, and ability to sense vibration all contribute to effective predation and scavenging. As the top predator in many freshwater ecosystems, the mobility and dominance of larger longfins increases their catchability relative to smaller eels. Larger individuals prefer pool habitats, which are the most productive in terms of food availability (Chisnall and Hicks 1993). This promotes their susceptibility to harvest, as the gear used by fishers is suited to placement here due to lower water velocities.

The feeding activity of freshwater eels is strongly related to water temperature, although they predominantly feed at night to avoid predators (such as the black shag (*Phalacrocorax carbo*)) and to increase the efficiency of their own predation. Longfins become less active and begin to hibernate in deep mud as the water temperature falls below 6°C. This usually occurs from mid-June to late August, although this depends on the locality of the waterway.

2.2.3 Natural mortality

Francis and Jellyman (1999) identified two estimates for instantaneous natural mortality (M) for eels above 40cm in length. These are .042 year^{-1} and .036 year^{-1}. These low levels of natural mortality arise from the role of larger longfins as the top predator in many freshwater ecosystems. The magnitude of these rates also indicates that there would have been little risk in delaying the onset of maturity to achieve increased levels of fecundity in the absence of commercial fishing (Jellyman 1995). This delay has played a key role in the disappearance of larger longfins from many freshwater ecosystems since the advent of sustained commercial harvest.

2.3 Reproductive biology

2.3.1 Recruitment

The longfin eel is a catadromous species, as after many years of residing in freshwater, it migrates to a part of the Pacific Ocean (believed to be east of Tonga) to spawn (Figure 2.3). Both males and females must make the spawning journey, although the male is often half of the size and age of the female at sexual maturity (Todd 1980) (see Section 2.3.2). Ovaries develop in females in the months leading up to migration, these containing between one million and twenty-one million eggs (Todd 1981a). Although no data on the survivability of young exists, it is postulated that the mortality of young is very high (due to predation, lack of food, and so on). This is a common trait among fish species with high fecundity (Beverton and Holt 1957; Jennings et al. 2001), for example in New Zealand hoki (*Macruronus novaezelandiae*) populations (Schofield and Livingston 1998).

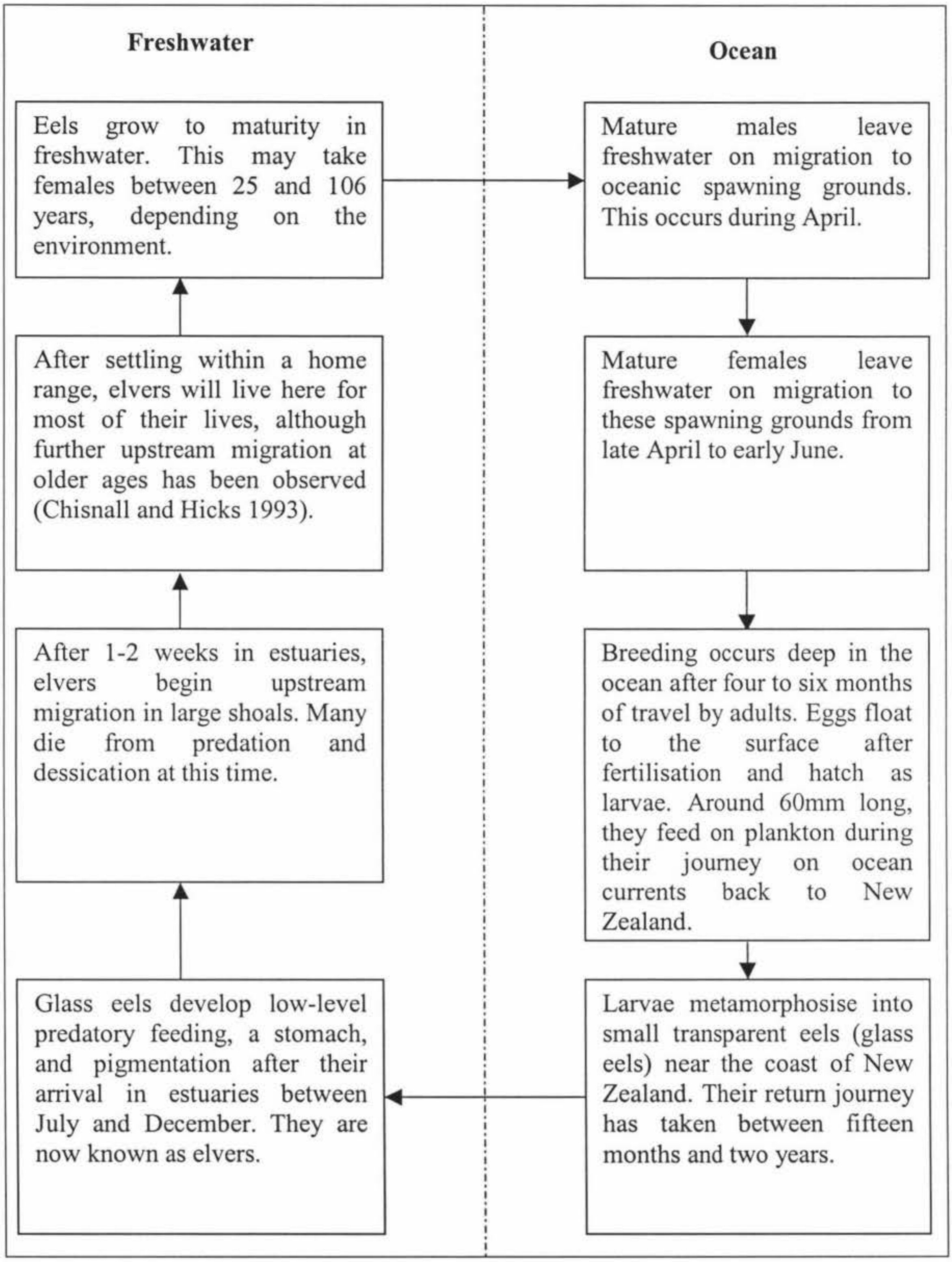

Figure 2.3. The reproductive cycle of the longfin eel.

Male and female eels breed at considerable depth, with eggs hatching as they rise to the ocean surface several days after fertilisation. Transparent, leaf-shaped larvae (leptocephali) then begin the journey on ocean currents back to New Zealand. It is estimated that this journey from the spawning grounds to New Zealand takes between fifteen months (Jellyman 1987) and two years (calculated from data in Jellyman and Todd 1982). Larvae are very vulnerable to predation and competition for food, due to a limited swimming ability, during this journey. Leptocephali transform into glass eels as they approach land. This metamorphosis involves the elongation of the body and the development of teeth, fins, and the ability to swim. At this stage the glass eels are around 60-70mm long. Glass eels generally enter freshwater between July and December and spend some time in estuaries and lower rivers before beginning an upstream migration (McDowall 1990). During this time they become pigmented and begin to develop several features that are characteristic of adult longfins. For example, these small specimens commence low-level predatory feeding while also developing larger stomachs and nocturnal habits. At this stage they become known as elvers.

Longfin elvers begin to migrate up waterways (together with shortfin elvers) in enormous shoals as they approach around 10 cm in length, primarily in January and February. These elvers are capable of moving across land and climbing any damp obstructions, including some hydroelectric dams. Such resilience has led to a wide distribution of longfin eels (McDowall 1990) (see Figure 2.2).

Once these small eels have completed their migration upstream, they will generally live in a specific area until they reach sexual maturity. For example, adult longfins inhabit a territory of less than 150m of a waterway's length (Chisnall and Kalish 1993). Eels may expand their territories into those of adjoining populations with low density, for example into those that have been exploited (Chisnall et al. 2002). This diffusion may increase their vulnerability to harvest.

The composition and concentration of glass eels entering into an individual waterway each year is highly variable, depending on the ocean currents that distribute them. For example, Jellyman et al. (2000) identified that over a five-year period at the Karapiro Dam on the Waikato River; the number of longfins in the elver migration could differ by up to 300 percent. The recruitment of elvers into a waterway is consequently either partially or totally independent of the stock of individuals reaching sexual maturity within it (Bardach et al. 1972). Annual variation in recruitment has complicated the identification of the true decline in longfin populations, especially since New Zealand lacks any time series of sufficient length to base inferences upon. Although various studies have been undertaken (Jellyman 1977, 1979), Jellyman et al. (2000) reported that samples were too variable and too few in number to provide clear evidence about trends in the arrival of longfin elvers.

2.3.2 Sexual maturity

Adult eel morphology changes dramatically at sexual maturity. The head becomes more streamlined and reproductive organs develop rapidly, while the eyes and pectoral fins also enlarge. Eels of both genders may begin to move down waterways towards the sea as early as November, but the key periods of migration for longfins are April for males and late April to early June for females. Heavy spawning aggregations may occur, such dense runs being stimulated by both high rainfall and certain stages of the lunar cycle. Migrant eels travel around forty kilometres per day and take up to six months to reach their spawning grounds.

There is high variability in both length and age at migration for both genders (Burnet 1969; Todd 1980). For example, Todd (1980) reported that the length of migrant males ranged from 482mm to 736mm (which corresponds to a weight range of 187.3g to 837.4g) after between twelve to thirty-five years in freshwater. Additionally, Todd (1980) identified that females migrate at lengths between 737mm and 1560mm (a weight range of 841g to 11.95kg) after around twenty-six to sixty years in freshwater, although Jellyman (1995) identified that it may take females up to 106 years to reach sexual maturity.

These ranges demonstrate a number of important factors. First, wide variability in both length and age at sexual maturity fail to provide any strong evidence on the primary determinant of maturation (Todd 1980; Francis and Jellyman 1999; Jellyman et al. 2000; Hoyle and Jellyman 2002). Second, sexual dimorphism is clearly evident given the smaller size and lower age of males at sexual maturity. The comparatively higher age of spawning females identifies their greater vulnerability to exploitation, even at low rates of exploitation, due to the cumulative impact of annual harvest (Hoyle and Jellyman 2002). Third, it is apparent that females spawn well above eighteen years of age, the age at maturity past which the probability of extinction for a fish species becomes considerably higher (Parent and Schriml 1995).

Sexual maturity at significant ages plays a central role in the overexploitation of longfin eels, as females may have to avoid harvest for up to eighty years in some environments before spawning. Capture during this time is promoted through the placement of gear in the preferred habitat of larger longfins (pools), the high mobility of adults, and the periodicity of fishing (sometimes up to two or three times at one site annually) arising from poorly defined property rights. Recruitment overfishing (whereby stocks are depleted to the level that recruitment declines significantly) (Clark 1985) is further promoted through semelparity, given that any fishing or pre-spawning natural mortality will always occur before reproduction (McCleave 2001; Hoyle and Jellyman 2002).

2.4 Growth rates

Longfin growth rates are highly variable across both habitats and time, but are generally very slow in wild fisheries (1-3cm a year) (Horn 1996). For example, Jellyman (1995) reported that the population in Lake Rotoiti (Nelson Lakes National Park, New Zealand) had the slowest recorded growth (.9cm year^{-1}) of any anguillid population in the world.

A linear relationship between age and length is commonly observed for longfins (Cairns 1941; Chisnall 1989; Chisnall and Hicks 1993; Horn 1996; Beentjes and Chisnall 1997, 1998; Beentjes 1999). This is supported by several studies that have identified a similar relationship for other anguillids, for example *A. rostrata* (Hansen and Eversole 1984) and *A. anguilla* (Vollestad and Jonnson 1986). This association is the converse to that described by many classical growth models for fish (for example, the von Bertalanffy relationship) (Quinn and Deriso 1999) because growth does not decline asymptotically as a maximum length is reached; this may arise from longfins not having to expend energy for spawning each year (Jellyman 1997).

Longfin growth rates are highly variable, with habitat effects being particularly important. For example, Chisnall and Hicks (1993) identified that the growth of longfins in waterways below indigenous forest was significantly lower than those in pastoral streams, reflecting differences in water velocities (and thus energy expenditure), food availability, and water temperature.

Growth rates may also be affected through exploitation. Chisnall and Hayes (1991) reported a positive relationship between levels of harvest and shortfin growth rates, as decreased densities led to greater food availability and consequently faster growth. Jellyman et al. (1995) identified a similar trend. Although both were limited to the study of shortfin eels, the underlying intuition provides some foundation for expecting that the same relationship could also hold for longfins. The existence of density-dependent growth for longfins is also suggested by the work of Chisnall and Hicks (1993), who found that lower eel densities in hydro lakes helped to promote faster growth. Additionally, Jellyman (1997) reported that competition for both food and space among longfins contributed to decreased growth rates. These findings suggest a relationship between longfin density and growth, which ideally should be accounted for in the determination of suitable regulatory strategies.

2.5 Conclusions

Slow growth and the reproductive biology of the longfin severely complicate its management. The long period of time required to attain sexual maturity is the primary cause for its high sensitivity to exploitation, given that female longfins may be exposed to commercial fishing for up to eighty years before leaving freshwater to breed. The probability of capture over this time is promoted through the periodicity of fishing (sometimes up to two or three times at one site annually), the placement of gear in the preferred habitat of larger longfins (pools), and the high mobility of larger females. Moreover, semelparity places this species at considerable risk of recruitment overfishing. It is subsequently apparent that sustained commercial harvest will be detrimental to longfin recruitment, consistent with the findings of Jellyman et al. (2000) and Hoyle and Jellyman (2002).

3. THE COMMERCIAL HARVEST OF LONGFIN EELS

3.1 Introduction

The management of the longfin fishery is complicated by the presence of a variety of user groups as commercial, customary, and recreational fishers all compete for a share of this resource. This thesis focuses on the commercial fishery, as market price is used as an estimate for the opportunity cost of both customary and recreational harvest. This follows from the primary role of commercial harvest in this specie's overexploitation (Jellyman et al. 2000), especially given the efficiency of gear and this user group's dominant role in exploitation,[2] and overcomes the ethical and practical difficulties associated with the estimation of non-market values.

First, in this chapter, the development of the commercial fishery is examined. The economic benefits accruing to this resource are then explored. This is followed by a description of the fishing techniques adopted by these fishers before the effects of commercial harvest on longfin populations are analysed. The management of the commercial eel fishery is then examined.

3.2 Development of the commercial fishery

In contrast to the intermittent attempts made at establishing a commercial fishery throughout the first half of last century, by the early 1970s a thriving export industry had developed for New Zealand's freshwater eel products. Harvests increased gradually from the mid-1960s, rising to a peak of 2434 tonnes in 1975 (Figure 3.1). Eels were the nation's second-most valuable fish export in that year (Jellyman 1993).

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[2] Ministry of Fisheries (2000) estimate that commercial harvest is responsible for 78 percent of total catch.

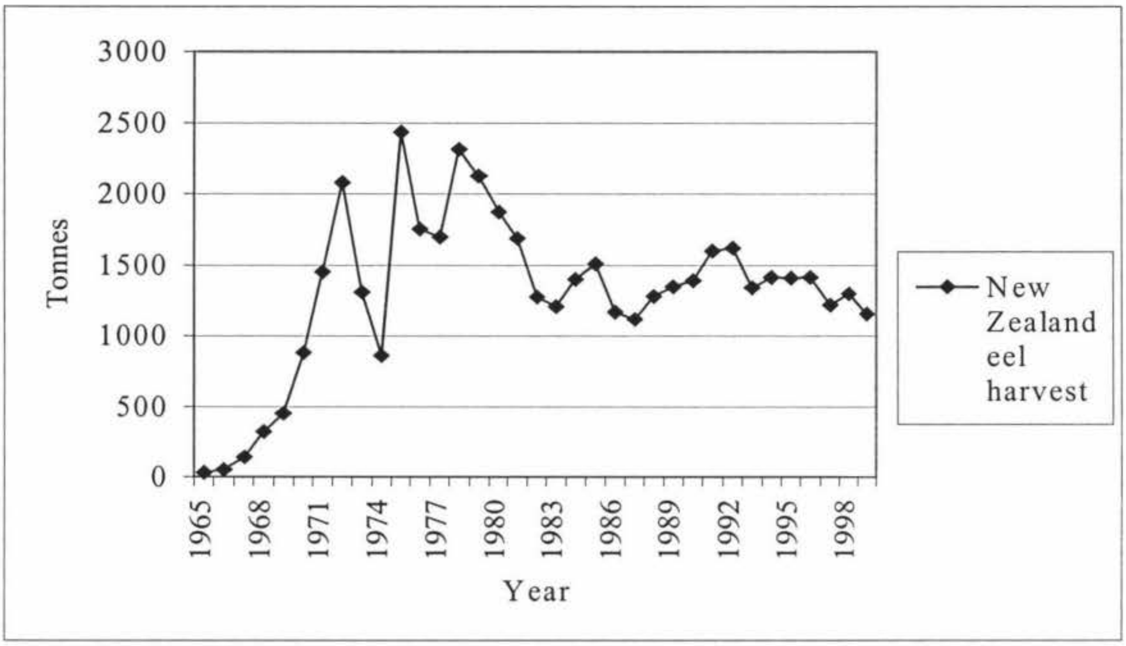

Figure 3.1. New Zealand eel harvest for 1965-99 (Annala et al. 2001).

The increase in harvest during the 1970s (Figure 3.1) reflects the presence of various important factors (McDowall 1990). Rising prices and low costs, low levels of regulation, and the attraction of eel fishing as a form of supplementary income increased the intensity of effort and the fishing of unexploited areas. Exploitation was also encouraged through the development and utilisation of superior technology, for example the introduction of fyke nets from Europe.

After experiencing dramatic fluctuations throughout the 1970s, harvests reached a relatively more stable trend in later periods (Figure 3.1), with yield fluctuating around a mean of 1457 tonnes since 1980. This represents the onset of regulation, for example the exclusion of part-time fishers in 1984, and the increasing difficulty associated with the location and harvest of unexploited eel populations. This pattern is observed in the initial stages of many fisheries, as high levels of cropping are followed by more stable rates as information increases, short-term fishers leave the industry as returns to effort fall, and management strategies evolve. However, these results for the freshwater eel fishery are misleading, as this stability fails to correctly portray the overexploited state of longfin populations.

The effect of historical harvest on longfin stocks is difficult to identify due to a lack of quality information. The detection of the specific proportion of longfins within annual harvests is complicated because around 40 percent of eels each year fail to have their species identified (Annala et al. 2001). This has prevented any meaningful estimates pertaining to the species composition of commercial harvest to be made since 1992 (Jellyman et al. 2000). Additionally, the data presented in Figure 3.1 represents the best estimate of annual harvest drawn from a combination of processor records, Ministry of Fisheries Licensed Fish Receiver (LFR) returns, Catch and Effort Landing Returns (CELR), and New Zealand Fishing Industry Board export data. Jellyman (1993) states that there are often wide differences between these measures of harvest due to the incompleteness of CELR returns, non-compliance, and/or the understatement of catch. Data distortion also occurs due to the incorrect classification of harvest by area. Stock assessment and the formation of biological reference points are also problematic, as longfins are widely distributed, experience highly variable growth between environments, and their limited home ranges prohibit the effective utilisation of mark-recapture methods (Ricker 1975) to estimate population strength (Annala et al. 2001). Furthermore, high variability in the recruitment of both species increases the complexity of identifying the impact of harvest.

An additional concern is the insufficiency of catch and effort data as regional figures vary widely between years, possibly due to inconsistency in recording. This prevents the use of this information for stock assessment; especially since species composition data is also of a low quality. The poor quality of catch and effort data is evident in national Catch-Per-Unit-Effort (CPUE) statistics for 1990-99[3] (Figure 3.2). Figure 3.2 illustrates that CPUE was relatively stable in the initial years of this period, but increased towards the end. This increase is counterintuitive as longfins have historically composed between 35 and 40 percent of total harvest, and this species is becoming increasingly overfished (Jellyman et al. 2000; Hoyle and Jellyman 2002).

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[3] A longer series is not available because a change in the unit used to measure effort, which was implemented in 1989, prevents the use of prior information.

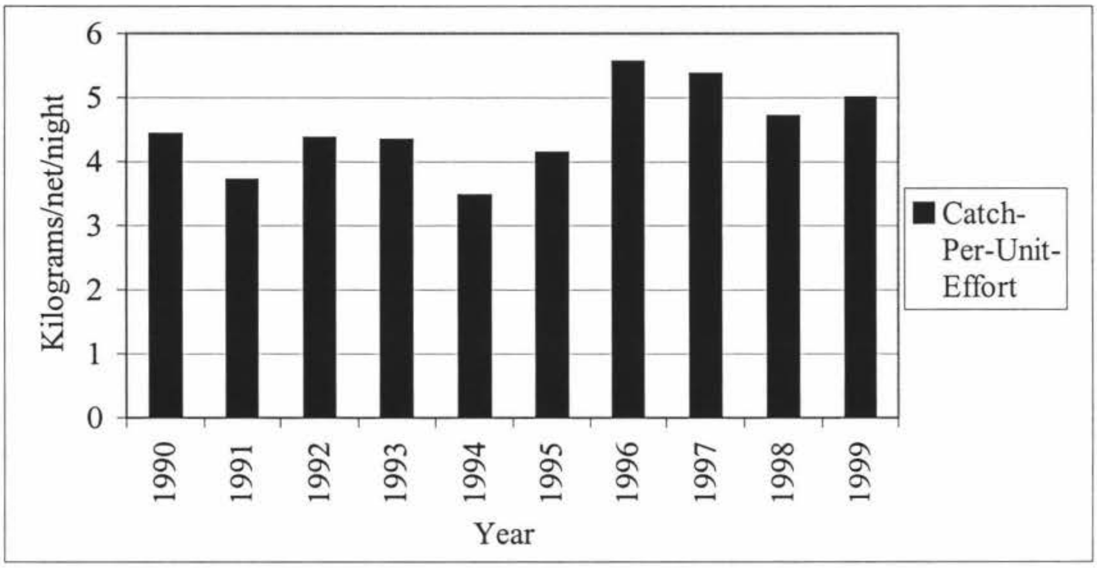

Figure 3.2. Catch-Per-Unit-Effort (kg/net/night) for the New Zealand freshwater eel fishery for 1990-99 (Ministry of Fisheries data).

Without information to provide insight into the relative strength of shortfin and longfin stocks, fishery managers have assumed that the populations of both species are stable based on recent harvest statistics (Ministry of Fisheries 2000). Yet the poor quality of data discussed above and the sensitivity of the longfin to exploitation (Jellyman et al. 2000; Hoyle and Jellyman 2002) challenge the validity of this assumption, especially since the longfin is more likely to become overfished due to its maturity at a larger size. For example, Todd (1980) identified that shortfin males reach sexual maturity at a mean age of 14.3 years, compared to longfin males that have a mean age of 23.2 years, while shortfin females reach sexual maturity on average at 21.8 years, compared to 41.9 years for longfins. The lower proportion of longfins in commercial catch and the recruitment of young shortfins from spawning stocks outside of New Zealand[4] also increase the difficulty of detecting a decline in longfin populations as stable harvests may be sustained for some time. Additionally, recruitment failure is difficult to identify in the short-term, given that yields may be maintained as young longfins take between five and thirty years to enter the fishery (Annala et al. 2001). The stability of harvest

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[4] Unlike the endemic longfin, the shortfin is also found in Australia, New Caledonia, Fiji, and a number of other Pacific Islands (McDowall 1990; Annala et al. 2001).

(Figure 3.1) and CPUE (Figure 3.2) in recent years provides an overly optimistic view of the sustainability of this fishery for these reasons.

These factors demonstrate the insufficiency of using past harvest as a benchmark for the calculation of quota and the subsequent need for greater insight into sustainable management strategies for the longfin. These features also identify the importance of improving data collection, as this will have potential benefits for management as the true status of stocks may become increasingly evident.

3.3 Economic benefits of commercial harvest

Eels have been a valuable export product in the past, being the second-most valuable fish export in 1975 and the fifth in 1980. Such high rankings are less common today, with eel exports comprising only .004 percent of total export earnings for New Zealand fish in 1999. This reflects the development of valuable deep-sea fisheries, such as that for the hoki (*Macruronus novaezealandie*) and orange roughy (*Hoplostethus atlanticus*), since the declaration of New Zealand's Exclusive Economic Zone in 1978. Nonetheless, the freshwater eel industry provided mean annual export earnings of around $4,758,478[5] between 1988 and 2000. Multiplier effects are also important. Te Waka a Maui me ona Toka Mahi Tuna (1996) reported that one study of the New Zealand fishing industry identified that for every dollar of output generated within it, $3.10 of output was generated elsewhere. It follows that between 1988 and 2000, the commercial eel fishery might have generated mean annual output within the economy of $14.75 million. These factors identify the importance of gaining insight into the sustainable harvest of longfins, as this species makes up a considerable proportion of a small, but significant, commercial fishery.

The large majority of harvest is exported, but there is an unknown (but reportedly significant) domestic market that is believed to consume between five and ten percent of processed product (Jellyman 1993). Eels are exported alive or are smoked, frozen, or chilled in whole or gutted form. Frozen eels dominate exports (Jellyman 1993;

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[5] This figure is estimated from Free-On-Board (F.O.B.) data provided by Statistics New Zealand and is specified in New Zealand dollars.

Statistics New Zealand data). Belgium is the principal importer of New Zealand's eel products, although Germany, the Netherlands, the United Kingdom, Italy, and France also play important roles (Table 3.1). Exports to the most important importers of these products (Belgium and Germany) decreased by 27 percent (Table 3.1) between 1998 and 1999. This reflects the periodic fluctuation of harvest illustrated in Figure 3.1.

Table 3.1. Principal nations importing New Zealand eel products (by weight), 1998-1999.

Nation	1998 imports (kg)	1999 imports (kg)
Belgium	270,120	186,206
Germany	151,090	121,687
Netherlands	73,834	40,698
United Kingdom	50,220	53,868
Italy	25,400	11,000
France	4000	12,632

All eel product is obtained from wild fisheries, as there are presently no eel farms in New Zealand. High capital and feed costs, reliance on the highly variable recruitment of glass eels and elvers for replacement stock, the metamorphosis of females to males at high stock densities (Beullens et al. 1997; Krueger and Oliveira 1999; Hoyle and Jellyman 2002), and the presence of various diseases have historically contributed to the marginal economic viability of these enterprises.

3.4 A description of commercial harvesting activity

The commercial fishing season differs between geographic locations, the seasons becoming longer in warmer regions. For example, some eel fishing is possible in the Far North for most of the year, while in southern regions commercial fishing primarily occurs from October/November through to early May. The species composition of harvest also differs according to geography. Annala et al. (2001) reported that shortfins dominated harvests in the North Island (with a mean value of 80.9 percent), while longfins dominated catches in the South Island (with a mean value of 62.8 percent) between 1990 and 1998. Although this data is of questionable quality (see Section 3.1), the existence of these regional differences is consistent with differences in habitat preference (see Section 2.2.1).

Although in the past migrating eels have been targeted due to their superior qualities for human consumption, Hoyle and Jellyman (2002) identified that migrating animals are now seldom exploited. The fishing of mature animals has become increasingly rare due to the introduction of a maximum size limit in the South Island in December 1995.

Restrictions are placed on the gear of commercial fishers, limiting them to the use of fyke nets (Figure 3.3), hinaki (a baited cage consisting of wire netting over a steel frame), or set nets. Jellyman (1993) reported that around 98 percent of eels caught by the commercial fishery are harvested using fyke nets. These are laid in rows along the margins of waterways and left overnight in accordance with the nocturnal habit of both shortfin and longfin eels.

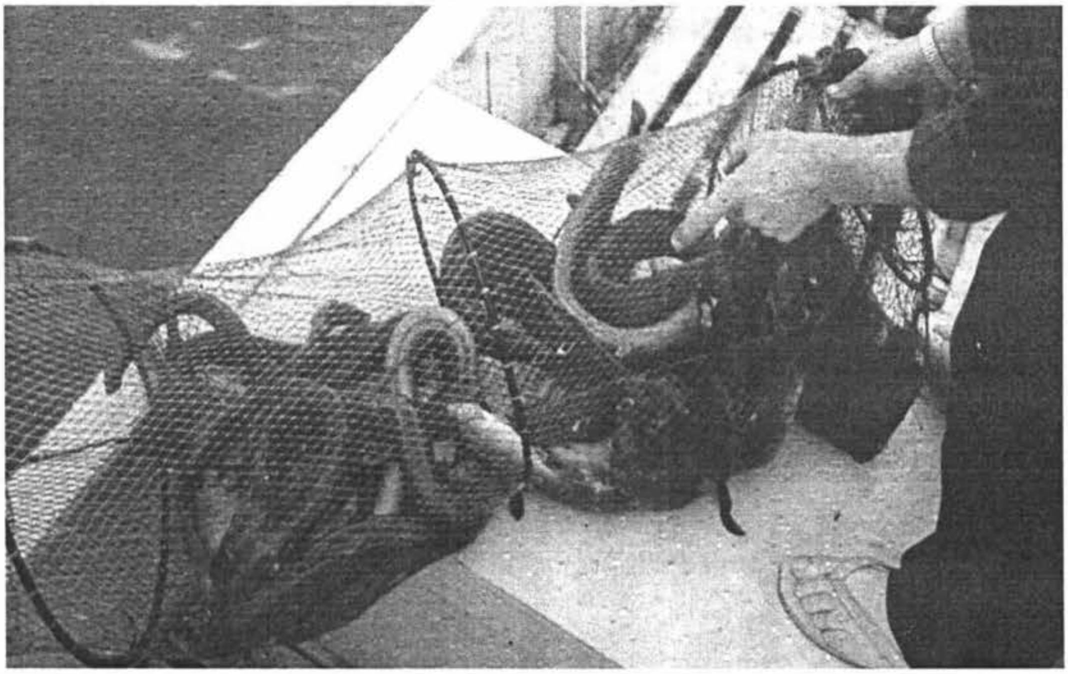

Figure 3.3. A small fyke net being emptied of its harvest (McDowall 1990).

Fyke nets have many benefits for the commercial fishery. Eels can be guided to them through the erection of a wing and/or through the setting of bait. These nets can be set in many freshwater habitats, except where swift water may clog them with debris or sweep them away. Their small size once collapsed, their adaptivity and flexibility, and the speed of their erection are also beneficial, contrasting the rather inefficient forms of equipment previously employed (such as hinaki) (Chisnall and West 1996; Jellyman et al. 2000).

Commercial eel fishing has other detrimental effects on freshwater fish populations, aside from its exploitation of eel stocks. Although by-catch of commercially important species using fyke nets is uncommon (Mathieson and Sutton 1981), rare native fish (for example, the short-jawed and giant kokopu (*Galaxias* spp.)) may be caught occasionally. Another concern is the spread of juveniles and eggs of nuisance species between waterways through their attachment to gear (Jellyman 1993). For example, the very aggressive koi carp (*Cyprinus carpio*) can be introduced into new habitats when equipment is not cleaned of their eggs. Levels of eel injury or mortality arising from harvest are not reported, but it seems that the number of affected animals is very low (M. Kuijten, *pers. comm.*, 1 May 2002).

Capital outlay and operating costs required for eel fishing are low, especially relative to ocean fisheries. The major items of expenditure are utility vehicles, small boats, boat trailers, and outboard motors.[6] High prices throughout the 1970s and various periods since have therefore stimulated profits and promoted the overexploitation of freshwater eels. The harvesting of virgin areas, where larger females are more common given the effect of exploitation on decreasing mean size within a fishery (Jellyman et al. 2000; Jennings et al. 2001), is also promoted through a price premium paid by processors for larger longfins (Figure 3.4). The data illustrated here is taken from telephone interviews with four of New Zealand's five major eel processors (see Section 6.3.1 for more information) and demonstrates that there is an economic incentive to harvest larger eels.

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[6] An overview of gear used by fishers is provided in a summary of the results of a survey of commercial harvesters in Question 15 in Appendix 3.

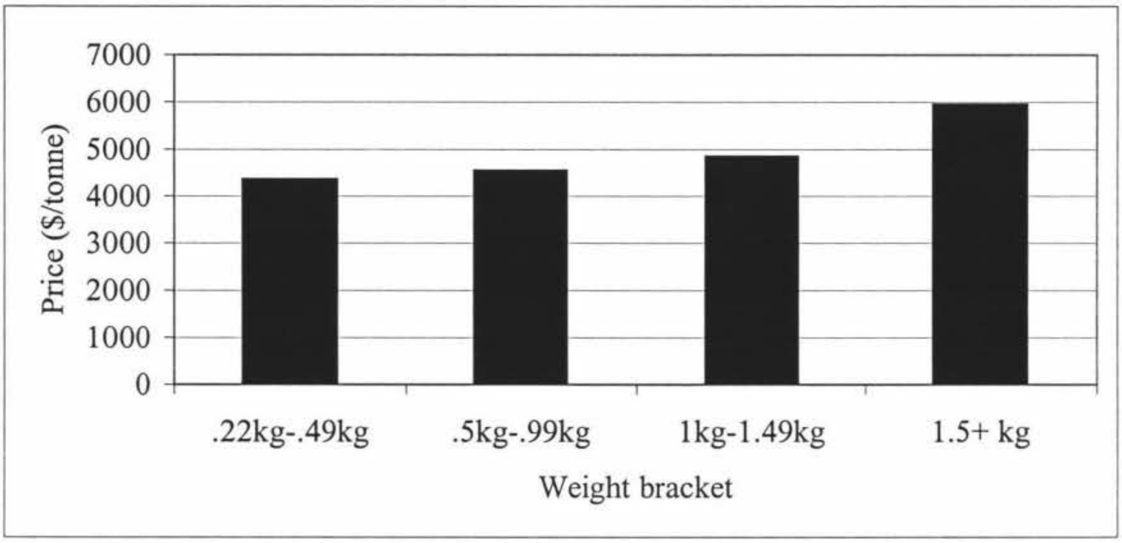

Figure 3.4. The mean price for individual eels within generalised weight brackets.

3.5 The effect of harvest on eel populations

Longfin eel populations are sensitive to exploitation due to their biology (see Section 2.3) and the effectiveness of modern fishing gear. Beentjes and Chisnall (1997) reported that historical processor data from both islands clearly indicated a progressive decline in the average size of processed longfins since commercial fishing began. Additionally, Jellyman (1993) stated that there was an obvious decrease in the average size of eel processed in one South Island plant between 1975-79 and 1984-90, with records showing a 300 percent increase in the smallest size category (.225 to .45kg). In accordance with these findings, it is apparent that commercial harvest has a detrimental effect on the size distribution of eel populations (Figure 3.5). The sample represented in the top figure was taken from an unfished site in 1948, while that in the bottom figure represents the same site almost 50 years later, after commercial fishing had occurred. The arrow signifies 75cm, which is around the minimum size required for the migration of a longfin female (Todd 1980).

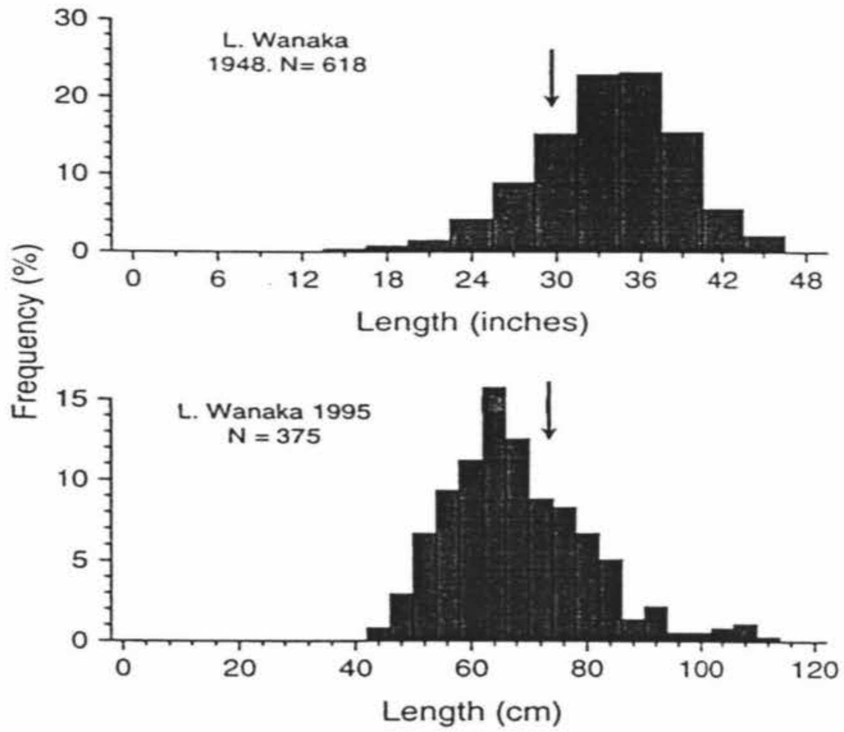

Figure 3.5. The effect of commercial harvest on a longfin eel population (Jellyman et al. 2000).

This data demonstrates the degree to which harvest can damage an eel population, as it is clear that a high number of larger animals have been removed. This has obvious ramifications for recruitment, given the presence of semelparity.

The effects of commercial harvest on the size distribution of longfin eel populations are also observable in regional harvests from both the North and South Islands (Beentjes and Chisnall 1997; Beentjes 1999; Chisnall and Kemp 2000). Only a limited number of longfin females between 600mm and the mean length of these animals at migration (1152mm, Todd 1980) are identified, demonstrating the detrimental effect of historical exploitation.

These findings validate those of Jellyman et al. (2000) and Hoyle and Jellyman (2002). Through the use of simulation modelling, these authors identified that even moderate rates of commercial exploitation will lead to great reductions in longfin numbers. Following assumptions that recruitment declined at 7 percent a year from 1980 onwards (to reflect the effects of historical harvest) and the population was subjected to moderate levels of fishing (10 percent a year for eels greater than 220g in weight) from

1970, Jellyman et al. (2000) identified that the number of females migrating annually decreased by around 90 percent after twenty years of exploitation. Additionally, Hoyle and Jellyman (2002) identified that exploitation rates of 5 and 10 percent would decrease spawning per recruit below unfished levels by 83 and 96.5 percent respectively.

It is clear from both harvest data and computer simulation modelling that sustained commercial harvest has a detrimental effect on longfin populations through decreasing spawning biomass and consequently recruitment.

3.6 Management of the New Zealand freshwater eel fishery

Although past regulatory strategies have had a number of benefits, these policies have been plagued by inherent deficiencies due to an apparent mismatch between their design and the characteristics of the problem. Additionally, the implications that ITQs, alternative weight restrictions, and area closures may have for economic efficiency have yet to be determined. These factors identify the need for analysing these policies in more detail.

3.6.1 Minimum weight restrictions

The current minimum weight restriction of 220g was established on 30 September 1993. This weight is implemented through minimum mesh sizes and the presence of at least two escape tubes in fyke nets. Minimum weight restrictions help to overcome growth overfishing (whereby the removal of small fish lowers total yield) through increasing Yield-Per-Recruit (YPR). This helps to minimise processing costs and maximise efficiency, yet YPR for female longfins is maximised at a minimum weight restriction at least twice the magnitude of the existing limit (Hoyle and Jellyman 2002). Additionally, a price that is increasing in individual size suggests that a higher restriction will promote efficiency. The existing limit also does nothing to overcome recruitment overfishing, which is the primary problem facing the longfin fishery. Generally speaking, the closer a minimum size restriction is set to the minimum size at sexual maturity, the more effective it will be in overcoming this problem. This identifies the insufficiency of the existing restriction as a measure to increase spawning biomass, as longfin females in many habitats will reach sexual maturity between twenty and fifty years after attaining this weight.

3.6.2 Maximum weight restrictions

A maximum size limit of 4kg was introduced for the South Island fishery in December 1995 to protect large female eels. The North Island is exempt from this restriction, primarily because more longfins are found in the South Island. Jellyman (1993) proposed the limit of 4kg, as although this excludes only half of spawning females from capture due to variability in size at maturity, fecundity information suggests that females larger than this weight would carry three times more eggs than those smaller (Todd 1981a). Eels above this maximum weight limit are removed from nets at capture and released, with evidence suggesting that the mortality of these animals on release is very low (M. Kuijten, *pers. comm.*, 1 May 2002).

The primary motivation for this regulation is the protection of those females that have reached sexual maturity and are moving downstream prior to oceanic migration (see Section 2.3.2). This restriction is particularly invaluable to protect spawners passing from areas where commercial fishing is prohibited (for example, national parks) through areas that are exposed to harvest (Chisnall and Hicks 1993; Jellyman 1993; Hoyle and Jellyman 2002) in the months preceding oceanic migration.

Lower maximum weight restrictions have been proposed. Chisnall and Hicks (1993) supported a maximum weight limit of 1.5kg, while Hoyle and Jellyman (2002) suggested that 2kg might be suitable. Despite this recommendation, the latter authors identified that a low maximum weight restriction would be difficult to implement in practice due to an economic incentive to harvest larger eels and handling mortality arising from their high catchability. These weight restrictions are also unable to protect eels that have yet to reach these limits, given the vulnerability of eels to the commercial fishery for many years before reaching them. For example, it may take a longfin female nearly twenty years to reach 1.5kg in a productive growth environment (pastoral site), compared to fifty years in less favourable sites (forested streams) (Chisnall and Hicks 1993). This indicates the general unsuitability of a maximum weight limit for preventing recruitment overfishing within a long-lived fish population, especially one that is semelparous. These restrictions might also impact on efficiency in a population in which price increases with size because a significant proportion of the most valuable eels would be unavailable for harvest.

3.6.3 Gear restrictions

Apart from the presence of minimum mesh sizes and escape tubes to enforce minimum weight limits (see Section 3.6.1), a variety of gear restrictions also seek to control the overexploitation of the freshwater eel resource. Commercial fishers may only use fyke nets, set nets, or hinaki traps. These must not be set within 60m of one another or set in such a way that more than a quarter of a waterway is blocked. Although these restrictions theoretically protect a population from very intensive exposure, the limited home ranges of freshwater eels, their high mobility, and the bait within traps mean that animals are often drawn to them anyway.

3.6.4 Restrictions on fishery size

The use of a permit system has sought to limit the size of the fishery. Following concerns about the fishery's sustainability throughout the 1980s, a variety of policies to this end were established. Part-time harvesters were excluded in 1984, decreasing pressure on the resource due to the significant number of people using fishing as a form of supplementary income prior to this. New permits ceased to be issued after 1986, being followed by a statutory moratorium on such issue between 1988 and 1991 (outlined in the Fisheries Act 1983) (Annala et al. 2001). Although the fishery was under brief regulatory control throughout 1991 and 1992, the national moratorium on the issue of new permits for species outside of the QMS (implemented in December 1992) led to long-term protection against increases in the number of commercial harvesters. There were 131 permits allocated to around 188 eligible fishers in the 1999/2000 fishing season (Annala et al. 2001).

Although limiting the entry of additional harvesters, this control has primarily been ineffective. Townsend (1990) reported that the effectiveness of these programmes is directly related to the level of restrictions contained within. In accordance with this finding, the lack of any sufficient controls on individual effort and harvest has allowed catch to remain constant or even increase. This continues through the use of past yields, which are consistent with causing severe recruitment overfishing (Jellyman et al. 2000), to calculate TACs.

3.6.5 ITQ management

ITQs allocate a fishing right to a harvesting entity, specifying a number/weight of a certain species of fish able to be harvested over a certain period of time in a specific place. The allocation of property rights theoretically mitigates the race-to-fish that characterises open-access fisheries as harvesters have an incentive to manage the resource sustainably.

ITQ management has many benefits for the regulation of fisheries. These systems are very flexible and are suited to the protection of a wide range of fisheries, including those with multiple species. A sustainable TAC will promote biological conservation, although this may be problematic due to uncertainty and environmental fluctuations. The allocation of property rights will also promote efficiency, given that decreased competition for fish will induce harvesters to decrease effort, fish over a longer season, and identify/use least-cost harvesting methods. In addition, the transferability of quotas will theoretically increase efficiency through the sale of catch entitlements by less efficient harvesters who cannot cover their variable costs. Fishers will also have increased financial security as TACs theoretically secure the sustainability of harvest. Furthermore, flexibility is promoted as catch entitlements are assets that may be freely traded, reducing the risk and uncertainty associated with investment in a fishery and permitting individuals to select their level of involvement. The integration of other conservative regulatory approaches with an ITQ system based on an appropriate TAC will also further protect a fishery from overharvest.

In addition, ITQ management has benefits for groups aside from commercial harvesters. Decreased fluctuations in harvest may benefit consumers as prices become more stable. Product quality and availability should also improve as quota holders attempt to match harvested product to market demand in order to increase profitability. Moreover, ITQ systems incorporate provisions for the inclusion of multiple user groups through the allocation of proportions of TAC among recreational, commercial, and customary harvesters. The transferable nature of quota also theoretically lowers enforcement costs, especially relative to input controls.

Nonetheless, ITQ management does have a number of primary limitations. Uncertainty and environmental fluctuations impact on the level of biological conservation offered by these frameworks, as sustainable TACs are difficult to identify. Unemployment may also follow the allocation and trading of quota, although this loss may bring significant long-term gains if the fishery is protected. The allocation of quota is also problematic, as it is difficult to distribute harvesting rights without causing significant inequity. The distribution of quota may therefore be costly and time-consuming, even within established systems, because of the differences between individual fisheries and the central role of fishing rights in determining the future income of harvesters. These initial difficulties are compounded through the social disruption that may occur during this process as conflict arises between fishers (Wilen 2000). Market power may also result from the trading of quota. Although specific limitations on quota ownership may be introduced to combat this, these result in efficiency losses if the most efficient harvesting scale is constrained. Harvesters will also retain an economic incentive to compete with other fishers if price premiums are given for certain animals, as less valuable fish are discarded and more valuable individuals are retained to maximise the value of quota (high-grading).

Although these limitations exist, the strong incentives for efficiency provided by ITQ management stimulated the formation of the New Zealand Quota Management System (QMS) in 1986. While the South Island freshwater eel fishery has been managed within the QMS for over two years, a lack of information and the short length of time since its inception present considerable barriers to gauging its effectiveness. The following discussion therefore concentrates primarily on the long-term theoretical benefits and limitations of ITQ management for the entire freshwater eel fishery.

Many analysts have supported the use of the QMS to protect the longfin eel resource (Town 1986; Te Waka a Maui me ona Toka Mahi Tuna 1996; Annala et al. 2001). While the benefits for efficiency and conservation outlined above would theoretically develop in this fishery, a number of additional advantages may also be observed. Profit-maximising harvesters may reduce pressure on longfin stocks by optimally targeting a *smaller* number of *heavier*, more valuable eels to fill a quota of a given size. This would only be beneficial if a sufficient number of individuals attained sexual maturity. This fishery may also benefit from a clearly defined process that is bound in legislation

(Te Waka a Maui me ona Toka Mahi Tuna 1996) and a system that has been continually refined over a significant length of time to increase its effectiveness as a regulatory instrument. Some cost may therefore accrue to alternative systems, given that fishery managers may not have experience in their design and implementation. A market-based quota system based on Annual Catch Entitlements (ACE) also decreases the need for expensive buy-back schemes and the inefficiency associated with non-transferable fishing rights. The allocation of a proportion of TAC to both customary and recreational harvesters also allows them continued access to this resource, although non-commercial harvesters are not allocated a quota *per se*, as these allocations are non-transferable.

Although a well-established ITQ management system is already present in New Zealand, a number of factors suggest that this method may be inappropriate for protecting the longfin fishery. The effectiveness of a quota system relies centrally on the accurate identification of an efficient level of harvest, but the specification of suitable TACs within this fishery is particularly problematic given a lack of information, especially in regard to stocks, fishing pressure, and variability in recruitment and growth (process uncertainty) (Caddy and Mahon 1995; Hoyle and Jellyman 2002). In addition, data quality may not improve under an ITQ system as fishers have a strategic incentive to under-report catch. Moreover, although poor information presents a considerable barrier to the objective formation of TACs, the stability of harvest fails to provide a true estimate of the strength of longfin populations (Section 3.2) and consequently the use of past catch to calculate TACs is inappropriate.

ITQ management may also fail to reduce competition among fishers. Competition would remain among commercial harvesters as larger eels are sought to maximise quota value (high-grading). Harvesters in all user groups would exhibit similar behaviour given that all prefer larger longfins (Jellyman 1993). This behaviour would promote the fishing of unexploited areas where larger eels are more common, decreasing spawning biomass and consequently affecting recruitment, especially since high TACs fail to provide limits on catch. In addition, competition between and within user groups for easily accessible fisheries (in order to lower travel costs) may prevent the recovery of populations in these areas, although low yields, because of past harvest and the sensitivity of longfin populations to exploitation (Jellyman et al. 2000; Hoyle and Jellyman 2002), should act as a deterrent to such behaviour.

Additional factors may prevent the successful protection of this fishery using ITQ management. These systems rely on significant net growth within a population to support sustainable and profitable exploitation, yet some species with low surplus production may not support a harvest level consistent with both. The longfin may be one such species because of semelparity, slow growth, and the time required for females to reach sexual maturity. Less efficient fishers may also become unemployed after the sale of quota, demonstrating the negative social effects that often accompany ITQ programmes (Symes 1999). These short-term losses in individual welfare are necessary for the long-term benefits of sustainable management to be realised, given the aggregate social costs of moratoriums in fisheries (Clark 1985).

ITQ management has many potential benefits for the regulation of the longfin eel fishery due to the incentives that the effective allocation of property rights have for the efficient utilisation of open-access resources. It is apparent that the successful application of these systems is problematic and therefore many of these benefits may not be observed in reality. The sustainable management of the longfin fishery presents a particularly difficult case for the use of ITQs, as higher prices for larger animals place pressure on those individuals that are approaching spawning. Moreover, significant uncertainty inhibits the accurate formation of TACs, the keystone upon which sustainability within ITQ systems ultimately rests. It is also possible that the use of annual quotas for the protection of the long-lived longfin is fundamentally flawed, as surplus production in these populations may be too low to sustain a profitable fishery because of slow growth and semelparity. These factors suggest that further insight into suitable management strategies for the longfin eel fishery is required.

3.6.6 Area closures

Area closures are a particularly promising method for the rebuilding and protection of spawning biomass within longfin populations (Chisnall and Hicks 1993; Jellyman 1993; Jellyman et al. 2000; Hoyle and Jellyman 2002), given the limited home ranges of these eels and high larval spillover (Sladek Nowlis 2000; Hoyle and Jellyman 2002). Extensive use of these closed areas would also have additional benefits. Fecundity may increase given that the size of maturing females will lift as the level of total exploitation falls (Jennings et al. 2001). Area closures may also ultimately increase longfin abundance and consequently permit higher harvest levels. Price may be unaffected by this abundance (given that downward pressure would be expected due to demand effects) as New Zealand's eel products have a wide range of close substitutes in the global market (McDowall 1990) and constitute only a small proportion of total supply. Most importantly, extensive application of this policy would overcome the difficulty of managing longfin fisheries to rebuild/maintain spawning biomass, given the high sensitivity of longfins to exploitation and significant uncertainty (Walters 2000; Hoyle and Jellyman 2002).

Current area closures are limited in the provision of these benefits (Jellyman 1993). While some are subjected to illegal fishing, 83 percent of the total protected lake and lagoon region within New Zealand has physical features that prevent the in-migration of eels (Annala et al. 2001). Chisnall and Hicks (1993) also identified that most waterways in these areas are beneath indigenous forest and far from the coast. Slow growth and low population densities are typical in these habitats (see Section 2.4), reducing their contribution to increasing spawning biomass. Spawners must also be protected as they pass through areas open to harvest, therefore maximum weight restrictions are necessary for maximising the spawning contribution of closed fisheries in inland areas (Chisnall and Hicks 1993; Jellyman 1993; Cadrin et al. 1995).

A number of limitations to the future use of area closures may also work against their successful implementation. Illegal fishing and the misreporting of catch may be difficult to prevent as harvesters have a short-term incentive to crop closed areas. Enforcement costs may therefore be high as non-compliance is difficult to observe. Additionally, area closures would affect the provision of fishing opportunities among users and regions through the displacement of existing commercial, customary, and recreational fisheries in productive waterways. The extensive use of closed areas would consequently have significant economic and social costs as decreased fishing among all user groups would lead to decreased earnings, utility, and prestige among customary fishers (mana). For example, closures would cause a number of harvesters to leave the fishery in the short-term due to an inability to cover fixed costs. Such losses among user groups may be significant, given that large areas may be required to provide for sufficient levels of escapement.

Biological limitations may also inhibit the successful implementation of area closures. The diffusion of eels into exploited populations after commercial fishing has occurred (Chisnall et al. 2002) may reduce the effectiveness of this regulation. This behaviour may render these closures entirely inappropriate, especially in regions where populations (and therefore competitive effects) are stronger (Chisnall and Hicks 1993; Jellyman 1997) and a waterway contains both closed and open areas. Additionally, the identification of suitable areas of sufficient size may be difficult since little is known of the reproductive biology of the longfin and spawner-recruitment relationships. The panmictic nature of longfins also suggests that populations in closed areas will decline as young eels disperse to areas that are freely fished (Jellyman et al. 2000). If elver or juvenile seeding strategies (Jellyman and Beentjes 1998) are subsequently used to promote closed populations, then not only will populations in open areas be denied this recharge, but closed areas will also suffer from the problems associated with higher densities in eel populations. The most notable are decreased growth (see Section 2.4) and the metamorphosis of females to males at high densities (Beullens et al. 1997; Krueger and Oliveira 1999; Hoyle and Jellyman 2002). Additionally, annual seeding would be difficult given sporadic recruitment and poor information.

The formation of management models for this fishery is also complicated because of a lack of information regarding longfin population dynamics (model uncertainty) (Fogarty et al. 1996). Since recommendations for area closure (Jellyman et al. 2000; Hoyle and Jellyman 2002) arise from modifications to a single model (Francis and Jellyman 1999), the presence of this uncertainty suggests that further insight from an alternative specification is warranted. This is important given that extensive adoption of area closures would have distributional effects. Additionally, none of these models have incorporated efficiency concerns or density-dependent growth (see Section 2.4) and thus do not include key considerations that may strongly influence recommendations for management.

Area closures are particularly suited to the protection of longfin populations, given considerable uncertainty, larval spillover, limited home territories, and the low probability of females reaching spawning under the present management system.

Nonetheless, these closures impact significantly on the provision of fishing opportunities among users and regions, while a number of biological characteristics, for example diffusion, further complicate their successful implementation.

3.7 Summary

The commercial fishery for freshwater eels is of both economic and social importance, given that it provides significant levels of export revenue and employment in both this industry and through multiplier effects. High levels of sustained commercial harvest since the early 1970s, arising from favourable price/cost ratios, technological advances, and ease of capture, have severely impacted New Zealand's longfin populations. This is consistent with modelling studies that have demonstrated that continued harvesting of a longfin population, even at moderate levels, will lead to a tremendous reduction in abundance, size, and spawning biomass (Jellyman et al. 2000; Hoyle and Jellyman 2002).

In accordance with these findings, various studies of New Zealand's eel harvest in both islands (Beentjes 1999; Chisnall and Kemp 2000) have identified the lack of large females in many exploited areas. This demonstrates the insufficiency of past management, because while weight and gear restrictions have failed to sufficiently protect spawning biomass (Chisnall et al. 2002), limits on individual effort or harvest did not exist for many years. While ITQ systems have many benefits, they may also fail to sufficiently protect longfin populations due to high existing TACs, slow growth, and a price incentive to target larger eels. While area closures have the capability to protect spawning biomass, these also have their deficiencies given problems with migration, diffusion, and distributional concerns. These factors identify the importance of gaining further insight into the optimal management of longfin populations.

4. THE OPTIMAL HARVEST OF MULTIPLE-COHORT POPULATIONS

4.1 Introduction

The first investigation of overfishing as an economic problem is frequently attributed to Gordon (1954) (Clark 1976a, 1985; Wilen 2000), who demonstrated in a static framework that poorly defined property rights are the primary cause of overexploitation and inefficiency within fishery systems. This was portrayed even more strongly through the extension of this model to include dynamic optimisation (Clark 1976a) subject to the characterisation of the problem in capital-theoretic terms (Scott 1955). Apart from displaying similar implications, both of these frameworks also based their representation of biological growth and decay (although only implicitly in the case of Gordon) on the logistic function popularised by Verhulst (1838), Pearl (1930), and Schaefer (1957). The powerful simplicity of this function has underpinned its popularity as the underlying biological specification of growth and decay within many bioeconomic studies that utilise lumped parameter frameworks (for example, Clark 1976a, 1985). Yet it is inappropriate whenever any biological characteristics, such as reproduction, are explicitly related to the age or size of an organism. It also involves the implicit assumption that recruitment occurs within the first period of a specie's life span (Clark 1976b).

It follows that the analysis of optimal harvesting strategies for cohort stocks has been based on alternative biological representations. One stream of literature has arisen from the portrayal of population growth in a matrix format (formulated independently by Lewis 1942 and Leslie 1945), in which a vector of a number of individuals (each element representing a single-cohort) is multiplied by a matrix (the so-called Leslie matrix) containing constant coefficients describing age-specific fecundity and mortality rates to obtain population levels in the next period. This formulation is inappropriate for the detailed study of many age-structured fisheries in its original form, given that linearities arising from its constant coefficients fail to cater for density-dependence and thus populations may increase exponentially under certain conditions. Nevertheless, it has been applied frequently in ecology (Usher 1972; Caswell 2001).

A second stream of literature has arisen from the seminal biological work of Beverton and Holt (1957), who developed a framework for static yield-effort analysis of an age-structured population with constant recruitment. This model has found wide use in both fish biology and bioeconomics due to a powerful simplicity that it shares with the logistic function. This model provides significant insight into the maximisation of yield in fisheries that match its restrictive assumptions and experience growth overfishing, given the ability to analyse alternative levels of age at first capture and fishing mortality. The difficult identification of stock-recruitment relationships is also avoided through assuming constant recruitment. This model's parameters are also biologically meaningful and can often be estimated from available data (Clark 1985).

Although providing a foundation for numerous analyses, this model in its original form had a number of important deficiencies. First, it did not incorporate density-dependence and thus prohibited the accurate portrayal of many fisheries. Second, this framework also failed to provide for the analysis of recruitment overfishing through its specification of exogenous recruitment, although this may be observed in an important number of fisheries that exhibit no apparent stock-recruitment relationship within some critical population range, for example prawns (*Penaeus* spp.) (Wilen 1985). Third, this model was cast in static biological terms and thus failed to account for time and economic considerations.

Investigations into the optimal harvesting of multiple-cohort populations have consisted of extensions to these seminal works, their combination in sophisticated analytical formulations, and the development/use of new methodology. This review follows the development of this area of research, primarily through an examination of analyses concerned with the determination of optimal harvest strategies for age-structured fisheries; nonetheless, other related topics, such as the investigation of size-structured populations, are also discussed if of relevance. Single-cohort analyses are included, as while these are not age-structured fisheries in the true sense, their focus on providing insight into the management of multiple-cohort systems, which are considerably more difficult to analyse, warrants their inclusion.

Two important factors are observable in the development of theory surrounding the optimal management of age-structured stocks. First, it is interesting to note how models have become increasingly complex, as mathematical techniques and computer technology have developed. Second, the factors underlying periodic harvest (the intermittent partial exploitation of a stock), bimodal exploitation (whereby a cohort is partially exploited in one period and then harvested totally in another), and pulse fishing (whereby the stock is completely removed in one period and allowed to grow before being completely exploited once again) are explored and compared in further detail throughout this review. The comparison of the origins and implications of these harvest profiles provides for increased insight into the optimal exploitation of multiple-cohort fisheries, as these are a general feature of many investigations.

4.2 Early analyses of the optimal exploitation of age-structured populations

Further investigation provided increased insight into the optimal economic management of fisheries following the pioneering work of Gordon (1954). Although not adopting the mathematical approach to modelling favoured by Beverton and Holt (1957), the static analysis of Turvey (1964) was based implicitly on this model through the use of similar model relationships, including the concept of a eumetric curve portraying the association between maximum yield and effort. Turvey (1964) demonstrated the importance of manipulating fishing effort and minimum mesh size to maximise economic yield from fisheries in which growth overfishing is observed. This work represented the first economic interpretation of the work of Beverton and Holt (1957), thus laying a foundation for further investigation involving temporal considerations.

Smith (1969) further developed the economic study of age-structured fisheries initiated by Turvey (1964) through the construction of an analytical framework that explored the economic effects of differing minimum mesh sizes, including a relationship between price and the size of fish at harvest. Although these considerations were ignored in this author's dynamic analysis due to the complexity that this would entail, this work provided the basis for further bioeconomic investigation utilising the Beverton-Holt framework through extending the work of Turvey (1964) to include mathematical relationships.

Before the use of the Beverton-Holt framework as a foundation for bioeconomic analysis using Optimal Control Theory (OCT) (Pontryagin et al. 1962), Goh (1972) utilised these methods to explore the optimality of alternative biological management strategies for a multiple-cohort fishery. This author focused on the maximisation of yield through the identification of optimal fishing effort and mesh size in a hypothetical fishery with perfectly selective gear. Although this analysis adopted the simplifying assumptions of Beverton and Holt (1957), this interpretation of their model laid the analytical foundations for its bioeconomic interpretation using OCT by Clark et al. (1973).

In contrast to the Beverton-Holt model, the study of optimal harvesting within the Leslie framework remained in biological theory for some time. Its extension by Leslie (1948) was primarily focused on overcoming several of this model's key deficiencies in regard to its use as a biological tool, including allowances for logistic growth and predator-prey relationships. This author also displayed that population structures would remain the same if mortality were independent of age, therefore providing the first insight into the analysis of harvest utilising the Leslie framework.

Lefkovitch (1966) was the first author to investigate exploitation within the Leslie framework in any detail, focusing on harvest profiles that may achieve a stationary population size. This analysis was weakened in that no consideration was given to the maximisation of yield and density-dependence arising from harvest. Nonetheless, the extension of matrix models to analyse harvested populations laid an important foundation for further work.

A number of computer simulations played important roles in the development of age-structured harvesting theory around this time. Walters (1969) demonstrated that maximising harvest involved intensively cropping reproductive age groups, as long as adequate spawning levels were maintained. This result resembles the bimodal harvesting policies that followed the optimisation of the linear Leslie model in later analyses (Beddington and Taylor 1973; Rorres and Fair 1975), as these also involved the heavy exploitation of reproductive cohorts. Additionally, Walters (1969) identified that periodic harvest may be optimal when age at first capture cannot be controlled. This followed the use of a dome-shaped Ricker stock-recruitment function (Ricker 1975), as periodic harvest provided higher levels of yield as losses in recruitment arising from density-dependence at high spawning populations were minimised. Moreover, Walters (1969) demonstrated the importance of non-selective gear and rapid growth at young ages (arising from the use of a von Bertallanffy growth relationship) to the optimality of periodic harvest. When that year class at which individuals obtain their maximum weight cannot be preferentially selected, Walters (1969) reported that it is optimal to heavily fish a population, lower its mean age, and consequently benefit from letting it grow rapidly. Pope (1973) reinforced this finding in a simulation study.

4.3 Development of harvesting theory within the Leslie and Beverton-Holt frameworks

While previous work had provided only limited insight into the optimal exploitation of age-structured stocks, extension of the Leslie and Beverton-Holt frameworks throughout the 1970s provided detailed theoretical foundations for future work. For example, Beddington and Taylor (1973) developed a more suitable framework for analysing harvest within the Leslie framework than that provided by Lefkovitch (1966), demonstrating the optimality of bimodal harvest through constrained maximisation of yield. This recommendation arose from specifications of linear physical growth and an implicit assumption of perfect selectivity within this and later variants of the classic Leslie model (Rorres and Fair 1975; Rorres 1976), as pulse harvest consequently had no benefit for increasing growth through the lowering of the mean age. The inclusion of stock-recruitment relationships is also pivotal, for as long as adequate recruitment levels were maintained through the partial exploitation of age groups with high reproductive contributions, it remained optimal to intensively fish older, less fecund cohorts.

Numerous extensions to the Leslie matrix/linear programming method developed by Beddington and Taylor (1973) (Beddington 1974; Doubleday 1975; Rorres and Fair 1975; Rorres 1976) shared the limitations of this original work. These analyses were primarily theoretical expositions with limited practical application, focusing on the characteristics of an equilibrium harvest policy in unrealistic models (Mendelssohn 1976). The abstract nature of these works meant that important considerations (such as stock externalities and costs) were often disregarded. Their reliance on linear programming also weakened their applicability, for while this permitted the specification of a well-behaved optimisation problem, it failed to provide for the incorporation of density-dependence. Additionally, Mendelssohn (1976) identified that their aim of maximising yield and harvesting to a stationary population size was not optimal as such a policy failed to allow for a growth in harvest arising from an increased population size, which was essentially unbounded due to the lack of resource constraints within these frameworks.

A parallel stream of literature that also evolved from the original work of Leslie (1945) shared this strong focus on theoretical investigations into the harvesting problem. Instead of seeking to identify the harvest profile that would maximise yield and stabilise the age structure of a population as done by Beddington and Taylor (1973) and others (Beddington 1974; Doubleday 1975; Rorres and Fair 1975; Rorres 1976), these studies were based on harvesting excess growth in every period regardless of age structure stability. These analyses originated from the work of Watt (1968), who identified that harvesting the total productivity of a population over time would yield the largest crop. This concept was formalised by Dunkel (1970). Mendelssohn (1976), however, succinctly demonstrated that such an incremental policy is not optimal, for it will always be dominated by a profile that allows the total harvest of the remaining population at some stage.

While optimisation models based on the Leslie method focused on the maximisation of yield throughout the 1970s, economic considerations were incorporated within the Beverton-Holt framework relatively early by Clark et al. (1973). This followed the extension of the original Beverton-Holt model in a number of economic analyses (Turvey 1964; Smith 1969; Bradley 1970) and an earlier optimal control model (Goh 1972). In contrast to the maximisation of biological yield as the goal of management (Beverton and Holt 1957), Clark et al. (1973) focused on the identification of the level of fishing effort that maximised discounted profit, for a single age group and multiple-cohorts. The consequent effect on model output was strong, with a 20 percent discount rate reducing mean yield per recruit by around 35 percent and age at first capture by nearly seven years, relative to the maximisation of sustainable yield.

This work was important for a number of reasons. First, it provided a strong mathematical representation of the key economic concepts for age-structured fisheries in a dynamic optimisation framework, extending earlier descriptions of key factors by Turvey (1964) and Smith (1969). Second, this work also mirrored the importance of discount rates for stock management demonstrated in a dynamic model based on the logistic function (Clark 1976a). Third, it also provided an analytical foundation for the conclusions of Walters (1969) and Pope (1973), who demonstrated that periodic harvest might be optimal under certain circumstances. Clark et al. (1973) identified that pulse fishing was optimal when fish were harvested at zero cost,[7] recruitment was constant, and gear had imperfect selectivity. Constant recruitment meant that stocks could be completely harvested without affecting the return of young, while imperfect selectivity permitted increased growth through the lowering of mean age as individual cohorts could not be preferentially targeted (see Walters 1969 and Pope 1973). While limited by the analytical complexity that increased realism would encompass, these authors provided the first rigorous development of a bioeconomic model for age-structured fisheries. This provided a foundation for many later studies (Stollery 1984; Wilen 1985; Schott 2001; Gallagher and Sylvia *in press*).

Waugh and Calvo (1974) extended the work of Clark et al. (1973) by incorporating positive costs and the calculation of an optimal time path for exploitation. This time path provided an interesting alternative to the findings of Clark et al. (1973), for it involved an increasing mesh size rather than a single optimal age. This arose from the exploitation of each cohort across several periods, rather than at one time, given that heavy harvest increased costs and lowered prices *ceteris paribus*. While demonstrating the impact that increased economic realism could have on policy recommendations, this work failed to provide the mathematical rigour, and thus the detailed exposition, provided by Clark et al. (1973).

Hannesson (1975) extended the analysis of Clark et al. (1973) through a comparison of sustained fishing and periodic harvesting strategies using a Beverton-Holt simulation model. This study was notable for a number of reasons. First, Hannesson (1975) reinforced the analytical conclusions of Clark et al. (1973) by demonstrating the optimality of pulse harvest in a numerical application incorporating positive costs. Second, Hannesson (1975) identified that although periodic harvest may remain as an optimal policy, sustained fishing could often bring similar levels of reward. For example, it was demonstrated that constant exploitation produced a present value less than 10 percent lower than that earned through intermittent cropping. This provides

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[7] It is a limitation of this research that a detailed analysis of the implications of positive costs was not done because of the complexity that this addition would introduce.

some confidence to managers of fisheries in which periodic harvest is not practical, as the more feasible notion of trying to maintain some constant level of exploitation may differ little in terms of overall optimality.

Clark and Kirkwood (1979) further extended the formulation of Clark et al. (1973) to include competing vessel classes exploiting a number of different prawn species. This extension demonstrated the flexibility of this format for bioeconomic analysis and reinforced its suitability for analytical investigation. A general lack of information and an assumption of linearity, which failed to allow for gear saturation and crowding diseconomies, limited the applicability of any findings.

Dudley and Waugh (1980) provided an alternative to abstract analytical solutions (Clark et al. 1973; Mendelssohn 1978; Clark and Kirkwood 1979) through incorporating significant detail in their discrete-time interpretation of the Beverton-Holt framework. This approach incorporated stochastic catchability, natural mortality, and annual recruitment, extending the ability of the Beverton-Holt model to provide insight into the optimal management of real fisheries. This application is particularly notable for its use of a computer to solve a model of significant complexity, as this approach contrasted the formulation of less comprehensive problems using analytical methods.

From these analyses it is apparent that the Leslie and Beverton-Holt frameworks developed along alternative paths over the 1970s. While those based on the Leslie method were constrained by methodological difficulties and a lack of realism, extensions to the Beverton-Holt model were significant at both the analytical and numerical level.

4.4 Overcoming the limitations of the Leslie and Beverton-Holt frameworks

In light of the limitations of the Leslie and Beverton-Holt frameworks, discrete-time models incorporating density-dependence and an explicit stock-recruitment relationship were increasingly utilised around this time. These analyses were logical extensions to the first incorporation of a non-linear stock-recruitment relationship within a Leslie framework by Allen and Basasibwaki (1974).

Getz (1979) utilised this approach to demonstrate that periodic harvest may remain optimal with the inclusion of a stock-recruitment relationship and positive costs. This suggested a weaker link between periodic harvest and the specification of a stock-recruitment relationship than that proposed by Clark (1976a), identifying the relative importance of imperfect selectivity. This finding indicated that it is impossible to formulate strict guidelines that determine the relative optimality of bimodal and periodic harvest profiles, as attempted by Clark (1976a), as each arises from the complex interaction of relationships portrayed within a specific model.

Reed (1980) further investigated the implications of incorporating a non-linear stock-recruitment relationship in a discrete-time model based on the classic Leslie framework. Together with Getz (1979), this work provided for more detailed analysis of age-structured fisheries through the incorporation of density-dependence. This represented a conscious move away from more theoretical and abstract analytical formulations (Beddington and Taylor 1973; Clark et al. 1973). Moreover, Reed (1980) demonstrated the optimality of a bimodal harvest strategy when compensatory stock-recruitment relationships are included within a non-linear model based on the Leslie method. Getz (1980) extended this through demonstrating that bimodal harvest is also optimal for any biologically feasible stock-recruitment relationship, both linear and non-linear. These results collectively exhibited that bimodal harvest may remain optimal within more realistic matrix models incorporating density-dependence.

Alternative approaches were also developed to incorporate increased realism within age-structured models. For example, Deriso (1980) developed a bioeconomic optimisation model for age-structured stocks with a delay between the birth of young and their recruitment into the studied population. This work extended previous investigations incorporating delayed recruitment within lumped parameter frameworks (Allen 1973; Clark 1976b; Goh and Agnew 1978). This model incorporated significant biological and economic detail relative to other investigations at this time, while also allowing for the estimation of parameters through the regression of catch and effort data. However, it did not permit the identification of the equilibrium age structure of harvest, in contrast to the frameworks of Beddington and Taylor (1973) and Conrad (1982).

Botsford (1981) demonstrated the optimality of pulse harvest in a complex model that incorporated density-dependent growth and recruitment. This recommendation followed the significant benefit accruing to the easing of density-dependent constraints associated with such an extreme harvest profile. This finding extended the work of Walters (1969), who identified a similar relationship in an analysis incorporating a density-dependent stock-recruitment relationship. Additionally, Botsford (1981) identified that pulse harvest is intuitively more favourable when harvest costs decrease with higher stock levels or only arise during the process of fishing.

Age-structured harvesting theory entered a new phase over the late 1970s and early 1980s as analyses moved away from the extension of the Beverton-Holt and Leslie frameworks and incorporated more complexity as developments in mathematical methods (Botsford and Wickham 1978; Deriso 1980; Getz 1980; Reed 1980; Botsford 1981; Gurtin and Murphy 1981a, 1981b; Stollery 1984; Brokate 1985) and the use of computers (Hannesson 1975; Dudley and Waugh 1980; Conrad 1982; Gatto et al. 1982) permitted the solution of more sophisticated problems. Moreover, stochastic (Mendelssohn 1978; Dudley and Waugh 1980; Horwood and Shepherd 1981; Reed 1983) and economic (Deriso 1980; Conrad 1982; Gatto et al. 1982) considerations were increasingly incorporated.

4.5 Investigations into the effects of price on multiple-cohort harvest profiles

Conrad (1982) extended the modified Leslie approaches of Allen and Basaibwaki (1974), Getz (1979), and Reed (1980) through the development of a generalised model incorporating density-dependent survival and a positive stock-related cost externality. This model was used to explore the implications of prices/fecundity that decreased/increased with age in a primarily numerical application involving the hard clam (*Mercenaria mercenaria*) resource on Long Island, New York. This analysis demonstrated that younger cohorts should be exclusively harvested while older animals should be protected for breeding. A measurement error was later identified in this work (Hsiao 1985; Conrad 1985), which altered the recommended strategy to involve harvesting this population to extinction.

Gatto et al. (1982) investigated the implications of higher prices for older fish in a deterministic simulation model applied to a small European eel (*A. anguilla*) fishery.

This study showed that for a given level of fishing mortality, the optimal level of age at first capture is higher when these prices are considered.

In contrast to those earlier investigations that utilised numerical methods (Conrad 1982; Gatto et al. 1982), Anderson (1989) explored the effect of price on harvest profiles using an analytical approach. Although the developed models were not solved completely due to their complexity, their construction demonstrated that as price varies with size, age at first capture becomes less crucial and the optimal range of harvestable cohorts becomes more important. Anderson (1989) therefore identified the intuitive importance of assuming perfect selectivity when price varies with age/size.

Moyle (1999) developed a small analytical model for Australian crocodiles, without the specification of functional forms, to investigate the implications of price premiums and breeding capacity in a population divided into non-reproductive and reproductive cohorts. This analysis identified that more valuable, non-reproductive age classes should be harvested and reproductive cohorts should be protected. This harvest pattern is more readily achieved here, rather than in fisheries, due to the nature of exploitation (hunting an amphibious animal).

A number of analyses have also identified the benefits of delaying the opening date of fisheries so that larger harvested animals might earn a price premium (Kellogg et al. 1988; Onal et al. 1991; Milliman et al. 1992). Nonetheless, a relationship between price and size/age may not determine the optimal harvest profile for a species, given that delayed harvest may lead to increased natural mortality, lower product quality (Larkin and Sylvia 1999), or decreased growth.

4.6 Incorporation of greater analytical complexity

Research using analytical methods based on the Beverton-Holt framework continued throughout the 1980s. While assuming perfect selectivity, Feichtinger (1982) extended the model of Clark et al. (1973) to include a stock-recruitment relationship and age-specific mortality. Feichtinger (1982) identified the optimality of a bimodal harvest profile, this result resembling those of earlier investigations that utilised the Leslie framework (Beddington and Taylor 1973; Rorres and Fair 1975; Reed 1980; Getz 1980). This reinforced the importance of perfect selectivity and the inclusion of a stock-recruitment relationship to the optimality of this harvest profile relative to periodic exploitation.

Spulber (1983) investigated the optimality of pulse harvest under assumptions of perfect and imperfect selectivity. This author identified that bimodal harvest was most favourable when gear was perfectly selective, as the preferential exploitation of older cohorts would increase the growth rate of young through decreased competition and cannibalism. Alternatively, a pulse harvest of all individuals above a critical age would be optimal with an assumption of imperfect selectivity. This outcome is less preferable to bimodal harvest, as the resulting equilibrium may not be stable due to an inability to sufficiently control cohort interaction. This reinforced the findings of Botsford and Wickham (1978), who had earlier identified a similar result. Spulber (1985) identified that bimodal harvest was also preferable to periodic harvest when a demand curve and an industry marginal cost function were included within this framework. These findings challenge researchers to identify practical policies that may earn rewards close to those accruing to these optimal strategies, as year classes cannot be individually targeted in many age-structured fisheries.

Stollery (1984) extended the limited analysis of multiple-cohort populations by Clark et al. (1973) through investigating the optimal harvest rate and mesh size for these fisheries. This analysis is particularly notable for its inclusion of a standard Schaefer production function (Clark 1976a) that previous age-structured analyses had not incorporated. Using parametric optimal control, Stollery (1984) demonstrated optimal equilibrium stock and harvest levels for different discount rates and cost considerations, although only simple scenarios could be analysed due to the complexity associated with identifying solutions outside of the steady-state. This appears as a general feature of more sophisticated analytical age-structured models. Stollery (1984) demonstrated that many of the bioeconomic principles (such as the sub-optimal nature of unregulated, open-access conditions) illustrated by the "standard" model based on the logistic function (Clark 1976a, 1985) also hold for the multiple-cohort fishery. Moreover, this work extended the analytical bioeconomic interpretation of the Beverton-Holt model (Clark et al. 1973; Clark and Kirkwood 1979) through the inclusion of stock-related costs.

Although no complete analytical solution to the multiple-cohort problem has been formulated, significant developments have been made since the publication of partial results by Clark et al. (1973), Botsford (1981), Feichtinger (1982) and Stollery (1984). Anderson (1989) constructed a sophisticated analytical model that provided valuable insight into the management of these populations, although its complexity prevented its complete solution. In addition, Horwood and Whittle (1986a) derived a dynamic programming equation for general fishery models, which Horwood and Whittle (1986b) used to demonstrate a method for the identification of a locally linear optimal control. This method is potentially applicable to many multiple-cohort problems. These authors postulated that the implications of the standard harvesting model (based on the logistic function) (Clark 1976a, 1985) would hold for these applications. This statement partially validated the hypotheses of Stollery (1984) and Clark (1985), who speculated that any general analytical solution to the multiple-cohort problem would differ little from that implied by the lumped parameter model.

Wilen (1985) used a graphical approach to explore the optimal exploitation profile for *potentially* overlapping cohorts when there is positive harvesting costs. This was an important starting point for later work in that, although no precise solutions were obtained, it laid the conceptual foundations for the extension of the analytical single-cohort model to the two-cohort case and therefore the explicit treatment of selectivity.

Schott (1999, 2001) extended this work through the solution of an analytical model that incorporated two cohorts and allowances for both perfect and imperfect selectivity. This research identified that both may be optimal under different conditions, primarily determined by the relative size of searching and gear congestion costs. For example, joint harvest through imperfect selectivity may be optimal when search costs are significant. Although this model extended theory to include endogenous selectivity, the practical implications of this study are limited in that fishers may seldom select between gears with perfect or imperfect selectivity.

Schott (2000) extended the single-cohort model of Clark et al. (1973), utilising both analytical and numerical techniques, through the addition of a Cobb-Douglas production function incorporating diminishing returns. This analysis demonstrated that a constant level of exploitation is optimal in this case, in contrast to the bang-bang result of Clark et al. (1973). This followed the inclusion of stock-dependent costs, as a large population is required to keep costs maintained at a reasonable level (Schott 2000).

Additionally, these analyses (Schott 1999, 2000, 2001) demonstrated that the optimal path of exploitation may begin when fish are growing either slower or faster than the discount rate when endogenous selectivity or diminishing returns are incorporated. This identifies the importance of considering more sophisticated multiple-cohort frameworks, as conclusions may be widely different from those implied by the single-cohort model of Clark et al. (1973), in which harvest is always delayed until the growth rate is smaller than the rate of discount.

While analytical models provide general insight into the optimal management of age-structured fisheries, these frameworks are restricted in their ability to portray multiple-cohort fisheries as the addition of age structure itself adds considerable complexity, often prohibiting the inclusion of additional factors (such as stock-related costs). Yet failing to consider these factors may lead to the formulation of incorrect policy, especially when single-cohort analyses are used to gain insight into the management of multiple-cohort fisheries (Wilen 1985; Schott 1999, 2000, 2001). Numerical applications may therefore hold more promise in the formulation of suitable management strategies for individual fisheries.

4.7 Use of numerical methods to study age-structured fisheries

In contrast to analytical specifications, numerical methods allow the incorporation of considerable detail in age-structured fishery models (Rowse 1995; Larkin and Sylvia 1999; Lee et al. 2000). Nonetheless, the solution of numerical models can be problematic. For example, the "curse of dimensionality", the exponential increase in computation time associated with additional state and decision variables in dynamic programming formulations (Kennedy 1986), may limit the effective description of multiple-cohort populations in these frameworks (Kennedy 1986, 1989). It may also be difficult to identify a global maximum when using Non-Linear Programming (NLP) (Cacho 2000). Moreover, holistic simulation models are often expensive to construct (Clark 1985), do not incorporate optimisation (although search routines may be included, Kennedy 1986), require a significant amount of quality information (Ludwig and Walters 1985), and are essentially flawed if they fail to accurately portray a given system (Clark 1985; Schnute et al. 1989a).

Aside from these limitations, significant insight has been obtained from numerical models. For example, Horwood (1987) used numerical optimisation and simulation techniques to investigate the dynamics of fishing mortality in a deterministic model incorporating concave recruitment and linear costs. This work extended the findings of Hannesson (1975) by concluding that the optimality of periodic harvest relative to sustained cropping is determined by the discount rate. Horwood (1987) speculated that as an increasing discount rate decreased the optimal age at first capture, pulse harvest helped to overcome the inefficiency of growth overfishing associated with imperfect selectivity.

Kennedy (1989) investigated the optimal recovery of a cohort fishery with depleted stocks and excess vessel capacity. Although the adopted solution method did not allow the optimality of pulse fishing to be investigated, it was conjectured that such a harvest profile would be less preferable if vessel capacity earned no return when fishing effort was at its periodic low level. While extending multiple-cohort investigations to incorporate the consideration of excess capacity, this work also demonstrated how multiple-cohort problems may be solved using dynamic programming, even with the significant barrier posed by the "curse of dimensionality".

Kennedy (1992) extended the solution technique of Kennedy (1989) to determine suitable TAC levels and harvest profiles for a multiple-cohort fish stock. Periodic harvest was identified as the optimal policy, although this changed with alternative parameter values and functional forms. Kennedy (1992) reported the existence of a negative relationship between the discount rate and the incentive to fish periodically, as the specification of a Schaefer production function (and therefore a positive stock-related cost externality) meant that a lower/higher discount rate would promote/dampen the incentive to fish periodically as the cost of limiting harvest until the population built to significant levels was decreased/increased. This finding reinforced the relationship between the discount rate and the relative optimality of periodic harvest identified by Horwood (1987). Additionally, Kennedy (1992) reported that periodic harvest would become less optimal with a reduced stock effect, quadratic adjustment costs, and a reduction in the absolute price elasticity of demand.

The complexity that numerical optimisation modelling, particularly NLP frameworks (Rowse 1995), can incorporate is also observable in additional analyses. For example, these techniques have been used to analyse multiple-cohort fisheries that involve a price that varies with age (Kellogg et al. 1988; Onal et al. 1991; Milliman et al. 1992), seasonal changes in the intrinsic quality of fish (Larkin and Sylvia 1999), multiple species (Lee et al. 2000), heterogeneous fishing vessels (Gallagher and Sylvia *in press*), and multiple management objectives (Mardle et al. *in press*). Game theoretic studies, utilising numerical simulation and optimisation techniques, have also been formulated to research the harvest and allocation of age-structured populations (Helgason and Olafsen 1988; Klieve and MacAulay 1993; Armstrong 1999; Kitti et al. 1999; Duarte et al. 2000). These have intuitively demonstrated that significant gains may generally be obtained from improved cooperation among harvesters, as in many single species fisheries (Clark 1985).

This discussion identifies that converse to analytical formulations, numerical applications may incorporate significant complexity when analysing age-structured populations. The level of detail that may be incorporated in simulation models is particularly significant, given that these may involve over 100 parameters (Grimm 1999). Chavez (1996) stressed the importance of these approaches in age-structured modelling, reporting a general need to incorporate more realism through the inclusion of greater detail.

Ludwig and Walters (1985) demonstrated the limitations of holistic modelling through a comparison of the complex model of Deriso (1980) and a simplistic Ricker stock-production framework. The more sophisticated formulation (that of Deriso 1980) failed to provide better estimates, even though it incorporated a greater amount of detail. This limitation is also observable in the analyses of Matulich et al. (1988a, 1988b). These factors identify the importance of matching the complexity of age-structured models to the quality and quantity of available information (Hillborn and Mangel 1997).

In light of the significant developments made in age and size-structured fish population modelling since this time (Schnute 1985, 1987; Fournier and Doonan 1987; Deriso and Parma 1988; Schnute et al. 1989a, 1989b), De Leo and Gatto (1995, 2001) developed sophisticated simulation models to provide insight into the bioeconomic management of eels within the Valli di Comacchio lagoons in Italy. This analysis demonstrated that a substantial improvement could be obtained through the restocking of elvers, reflecting the exploited state that these lagoons share with many fisheries for *A. anguilla*.

A number of numerical analyses have also been based on close variants of the original Beverton-Holt and Leslie frameworks. The dynamic Beverton-Holt model (Clark et al. 1973) has been used in numerical studies that investigate conceptual problems (Schott 1999, 2000) and the management of fisheries that match its restrictive assumptions (Canedo-Lopez et al. 1999; Gallagher and Sylvia 2002 *in press*). Variants of the classic Leslie framework have been particularly useful in specific applications (Forsberg 1999; Lang'o and Owino 2000) to populations that are not self-regulating, for example in aquaculture where constraints imposed by environmental factors are less pronounced (Forsberg 1999). Together with the use of concepts identified by Beverton and Holt (1957) and Leslie (1945, 1948) in many formulations (Clark et al. 1973; Stollery 1984; Conrad 1982), this demonstrates the lasting effect that these seminal analyses have had on age-structured modelling.

4.8 Summary and conclusions

A number of important findings are evident in investigations of the optimal harvest of age-structured populations. One of these is the nature of harvest profiles identified by modelling, for throughout the literature there are numerous examples of both bimodal exploitation (Beddington and Taylor 1973; Rorres and Fair 1975; Rorres 1976; Feichtinger 1982) and periodic cropping (Clark et al. 1973; Hannesson 1975; Mendelssohn 1978; De Klerk and Gatto 1981; Spulber 1983; Horwood 1987; Kennedy 1992).

Periodic harvest may be practical in a number of real fisheries, for example where marginal fishing costs are decreasing in harvest.[8] Yet this pattern of exploitation is often impractical due to the loss of markets, employment, investment, specialisation, processing capacity, and gear effectiveness (if equipment is non-malleable) (Botsford 1981; Kennedy 1992). Bimodal harvest is also difficult to implement in many fisheries

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[8] For example, where large factory vessels and a high number of accompanying fishing boats heavily exploit an area until moving to a new location (Clark 1976a).

given selectivity constraints and uncertainty regarding year class strength. These factors identify the importance of using recommendations for bimodal and periodic harvesting strategies as yardsticks to which more practical policies may be compared (Hannesson 1975; Getz 1979, 1985), rather than as strict proposals for management. The validity of this approach is reinforced by the small differences in yield or profit that may be observed between constant exploitation and periodic harvest (Pope 1973; Hannesson 1975).

Recommendations for bimodal and periodic harvest profiles are the intuitive outcome of relationships incorporated within models of age-structured fisheries. For example, while periodic harvest is favourable when constant recruitment is assumed (Clark et al. 1973) and imperfect selectivity prevents the preferential harvest of slower growing older fish (Pope 1973), bimodal exploitation is also intuitive when reproductive year classes pass some optimal breeding age and may be specifically exploited (Beddington and Taylor 1973; Feichtinger 1982). Yet, while recruitment and selectivity assumptions obviously play important roles in the determination of the nature of harvest profiles (Clark 1976a), this review has demonstrated that exceptions exist. For example, periodic harvest may occur with the inclusion of a stock-recruitment relationship (Getz 1979). The many factors that influence the optimality of alternative harvest profiles identify the sensitivity of policy recommendations to model construction. This reveals the importance of rigorous sensitivity analysis and reinforces the utilisation of these conclusions as guides for more realistic policy. In addition, the importance of carefully interpreting model results is emphasised, especially since the true model underlying the population dynamics of a species is seldom, if ever, known (Fogarty et al. 1996).

This review has identified that prices that vary with individual size/age may have a profound effect on harvest profiles, as exploitation is concentrated on the most valuable cohorts (Gatto et al. 1982; Anderson 1989; Moyle 1999) and reproductive age groups are protected (where the year classes belonging to each diverge) (Moyle 1999). This demonstrates the importance of using bioeconomic models to guide the management of age-structured fisheries, as the consideration of price differences across year classes, discounting (Clark et al. 1973), and costs (Conrad 1982) often have a significant effect on recommendations for management (Clark 1976a, 1985; Anderson 1989).

It is also apparent that over time the theory underlying the optimal management of age-structured fisheries has moved away from abstract, conceptual formulations (Clark et al. 1973; Beddington and Taylor 1973; Rorres and Fair 1975) towards the utilisation of sophisticated numerical approaches (Larkin and Sylvia 1999; Lee et al. 2000; De Leo and Gatto 2001). With the loss of some generality, this has allowed greater insight into the management of specific fisheries as analyses have provided for increased realism. This is demonstrated in that while a number of sophisticated numerical analyses have utilised NLP to solve models involving age-structured fish populations (Larkin and Sylvia 1999; Lee et al. 2000), no general analytical solution to the multiple-cohort problem has yet been developed to the author's knowledge. However, the solution of numerical models can also be problematic. Holistic simulations are expensive to construct (Clark 1985), require a significant amount of quality information (Ludwig and Walters 1985; Hillborn and Mangel 1997), and are essentially flawed if they fail to accurately portray a system (Schnute et al. 1989a). Additionally, users of dynamic programming models must overcome the "curse of dimensionality" (Kennedy 1986), while the identification of global maxima may be difficult when using NLP (Cacho 2000).

The development of age-structured theory reflects the utilisation of developments within both mathematics and computer technology. After the initial investigations of Leslie (1945, 1948) and Beverton and Holt (1957), the analytical complexity of these formulations was significantly furthered using more advanced mathematical techniques. The development of OCT (Pontryagin et al. 1962), the basis of the Beverton-Holt model in fish biology, and the amenability of this framework to analytical calculus have promoted its extensive use for bioeconomic research (Clark et al. 1973; Stollery 1984; Schott 1999, 2001). While OCT (Reed 1980) and dynamic programming (Mendelssohn 1978; Getz 1985) were also applied to extensions of the Leslie approach incorporating harvesting (Beddington and Taylor 1973; Rorres and Fair 1975), the comparative ease of using the Beverton-Holt model for exposition has favoured its use over discrete-time formulations. The complexity that remains in determining the optimal management of multiple-cohort populations within analytical analyses has promoted the use of computers to solve numerical problems incorporating greater detail (Dudley and Waugh 1980; Conrad 1982; Larkin and Sylvia 1999; Lee et al. 2000; De Leo and Gatto 2001). This is consistent with the rapid development of computer technology that permits the solution of models of increasing sophistication (Deriso and Parma 1988; De Leo and Gatto 1995).

Even with this development, poor information and the complexity involved with portraying age structure within bioeconomic models have generally constrained progress within this area of research. Further work is therefore required, for example in the analysis of multiple species considerations, intrinsic product quality, industry capacity, and patterns of investment. The advancement of mathematical theory and computer technology holds promise in that modellers may increasingly use these tools to gain further insight into these problems through the incorporation of increased complexity in management models. However, this development is conditional on the further development of data collection processes, given that these continue to constrain many investigations in fisheries management.

5. THE ANALYTICAL MODEL

5.1 Introduction

While many analyses have investigated the optimal harvesting of age-structured stocks, existing models are unsuitable for describing the complex biology of longfin eels. This chapter focuses on the development of an appropriate model for investigating the optimal management of this fishery based on past literature and the features of this problem. Although the complexity of this framework prevents its complete analytical solution, its partial solution provides some insight into the characteristics of an optimal strategy, laying the foundation for the interpretation of results obtained from optimisation of the numerical model.

First, in this chapter, the past literature underlying the developed model is described. The analytical model is then developed, before it is partially solved to distinguish key features of an optimal solution.

5.2 Theoretical foundations of the model

Existing age-structured models are not suited for the investigation of this problem. Deriso (1980) developed an optimisation model that could describe semelparous populations with delayed recruitment, but this framework did not permit the study of individual cohorts, which is required in this case because price varies with individual size (Anderson 1989). Additionally, while many authors have conducted analytical investigations involving the Beverton-Holt model (Beverton and Holt 1957) (Clark et al. 1973; Stollery 1984; Schott 1999, 2001), their assumptions of constant recruitment and density-independent growth are not applicable here. While constant recruitment may avoid the problems associated with identifying an appropriate spawner-recruitment relationship, this approach is inappropriate for the longfin given the need to represent recruitment overfishing and the key role that dynamic adjustment in spawning biomass would play in the recovery of this species.

Other methods of depicting this problem also have their limitations. In accordance with the need to match model complexity to the quality and quantity of available data (Ludwig and Walters 1985; Hillborn and Mangel 1997), the sophisticated stochastic models developed for an Italian eel fishery by De Leo and Gatto (1995, 2001) are unsuitable due to poor information. Dynamic programming formulations, such as that used by Kennedy (1992), are also limited in that the "curse of dimensionality" prevents the depiction of the high number of harvestable cohorts present in longfin populations. For example, twenty fishable year classes are depicted in the base numerical model presented in Chapter 6.

The complexity of longfin biology also hampers the application of a number of analyses that have incorporated price differences across size/age classes. The complexity of the problem prevents the use of analytical methods to satisfactorily investigate the suitability of alternative management strategies (Anderson 1989; Moyle 1999), while an optimisation approach is favoured over the deterministic simulation model of Gatto et al. (1982), given that this formulation did not identify an economic optimum, assumed constant recruitment, and did not incorporate positive costs or density-dependence.

The limitations of these approaches motivate the adaptation of the generalised multiple-cohort model developed by Conrad (1982). This framework is a discrete-time age-structured optimal control model, incorporating a delay-difference equation to express a lag between the birth of young and their vulnerability to the fishery. The numerical model presented within this thesis extends the complexity incorporated in Conrad's specification, given significant developments in mathematical programming algorithms (Brooke et al. 1988; Drud 1985, 1992) and computer technology since this work was done[9] and the need to portray the biology of the longfin. First, the linear stock-recruitment relationship utilised by Conrad (1982) is replaced by a density-dependent specification. Second, density-dependent growth for harvested cohorts is included. While the model of Conrad (1982) incorporated density-dependent survival, this was omitted from this author's numerical application due to a lack of information. Third, the biological model is adapted to describe a semelparous population with sexual

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[9] This follows a general trend towards the incorporation of greater detail in multiple-cohort fishery models using non-linear programming by many authors (for example, Larkin and Sylvia 1999; Lee et al. 2000; Schott 2000).

dimorphism and delayed recruitment. Fourth, the steady-state analysis of Conrad (1982) is extended to include the analysis of approach paths. Fifth, the size of the model is significantly increased, from eight harvested cohorts to twenty in the base case. Sixth, the model is adapted to portray alternative weight restrictions and quota levels. Last, the pattern of pricing is reversed to cater for a price that increases with age.

Together with the identification of suitable regulatory strategies for the recovery and management of the longfin fishery, these extensions represent a valuable contribution to the general theory of the optimal harvesting of age-structured stocks for a number of reasons. First, they increase the realism incorporated within the general framework of Conrad (1982) due to the inclusion of two important density-dependent specifications. Second, the addition of delayed recruitment and semelparity provide for greater insight into the optimal management of age-structured populations that have these characteristics. Third, this model extends previous research through analysing the implications that a price that increases with size/age has for the management of a semelparous, long-lived, multiple-cohort population. Last, this model increases the flexibility of the general model of Conrad (1982) through extending this analysis to incorporate alternative minimum and maximum weight restrictions. The resulting specification has considerable appeal for application to other fisheries, given its amenability to mathematical programming and ability to portray multiple management scenarios.

5.3 Key assumptions

This model is based on a number of important assumptions. These are presented here, together with the system used to denote cohort references. These references are based on a range of year classes, $k=[0,\dots,k^s]$, where $k=0$ represents the elver stage and $k=k^s$ is the eldest age group, the female spawning cohort. The harvestable cohorts within this range are denoted as $k=[k^h,\dots,k^H]$, where the year class consistent with the minimum harvestable weight is denoted as k^h and that corresponding to the maximum harvestable weight is k^H.

The key assumptions underlying the specification of this model are:

1. It is assumed that no model relationships vary randomly over time. A deterministic approach is favoured as the addition of stochasticity may fail to provide any richer insight into the problem (Ludwig and Walters 1982; Clark 1985; Conrad and Clark 1987) and would require more accurate information regarding the probability distributions of model relationships, primarily recruitment.
2. While multiple species considerations are important because of significant habitat overlap between the shortfin and longfin (McDowall 1990), this analysis follows Jellyman et al. (2000) and focuses solely on the longfin. This provides increased insight through the incorporation of greater detail, most notably density-dependent growth and spawner-recruitment relationships that have yet to be included in management models of this fishery. Additionally, the development of this model provides a foundation for future analyses incorporating multiple species considerations.
3. The stock is studied in an aggregate form due to a lack of quality information (Ludwig and Walters 1985; Schnute 1987; Hillborn and Mangel 1997; Hoyle and Jellyman 2002). It follows that the members of each age class are all treated as "average" animals that have uniform size, catchability, growth, mortality, and patterns of sexual maturity in this model. While this is obviously a simplification, it allows for an analysis of the dynamic implications of given management strategies that biological analyses (Jellyman et al. 2000; Hoyle and Jellyman 2002) of the longfin fishery have yet to incorporate.
4. All stock and harvest variables are presented in tonnes; this aids the interpretation of output and provides a consistency that aids the formulation of the model.
5. It is assumed that vital parameters are related to both age and size. This follows from the classification of animals in each age class by a mean weight and length. For example, male eels within this model reach sexual maturity at the ages consistent with the range of lengths at which they have been reported to migrate (482-736mm) (Todd 1980). This approach represents a compromise between the contrasting views of the determinants of sexual maturity for the longfin, as while the most comprehensive analysis of migrating New Zealand freshwater eels reported that age is of primary importance (Todd 1980), many authors have identified the significance of length (Francis and Jellyman 1999;

Jellyman et al. 2000; Hoyle and Jellyman 2002) (see Section 2.3.2). This approach also permits a more efficient description of the multiple-cohort nature of longfin populations within a dynamic optimisation model, as this approach circumvents the complexity associated with using size to describe the population, as apparent in the work of Botsford (1981).

6. It is assumed that all harvestable cohorts ($k=[k^h,...,k^H]$) have a constant rate of catchability. Although larger eels are believed to be more prone to gear (Hoyle and Jellyman 2002), a lack of information prevents its inclusion.
7. It is assumed that perfect selectivity exists. This assumption provides richer insight into the effects of price on the harvest of individual cohorts (Anderson 1989) and the comparison of policies incorporating alternative weight restrictions.
8. Since the limiting factor for longfin recruitment is the availability of females (Jellyman et al. 2000; Hoyle and Jellyman 2002), the stock-recruitment relationship incorporated within this model involves only mature specimens of this gender. This follows the extensive use of this approach in matrix modelling (Caswell 2001). All females are assumed to mature at the age at which length-at-age is closest to that corresponding to the mean size of migrant female longfins (1156mm) (Todd 1980). This spawning year class (k^s) is not harvested in this model to reflect that migrating females are seldom exploited (Hoyle and Jellyman 2002). Although having all females migrate at one age is clearly an abstraction, the use of a mean value maintains some realism while also simplifying the estimation of a stock-recruitment relationship, the portrayal of semelparity, and the description of the long life of females.
9. The migration of males at sexual maturity is portrayed in a way that is synonymous with natural mortality. This permits it to occur over a wide range of ages. This approach follows the supposition that sufficient males are available for spawning, but females are not (Jellyman et al. 2000; Jennings et al. 2001).
10. The population is divided into two collections of year classes, juveniles and adults. Juvenile year classes ($k=[0,...,k^{a-1}]$) differ from adult cohorts ($k=[k^a,...,k^s]$) through dimorphism in dietary preference, as eels below 45cm in length (juveniles) feed from the substrate of their resident waterway, predominantly eating insect larvae and molluscs, while larger longfins (adults)

are predominantly piscivorous (Jellyman 1996, 1997) (see Section 2.2.2). Since competition for food and space has a strong effect on the growth rates of larger eels (Chisnall and Hicks 1993; Jellyman 1997), their growth within this model is classified as density-dependent, while that for juveniles is not.

11. It is also assumed that juvenile cohorts are never harvested. Eels around the existing minimum weight restriction (220g) are approximately the length of those in the first adult age class (45cm),[10] therefore strong recommendations for increasing existing restrictions (Chisnall et al. 2002; Hoyle and Jellyman 2002) implicitly specify that juveniles should never be exploited. It follows that the year class corresponding to the minimum weight limit (k^h) will always be the same age as the first adult cohort (k^a) or older; that is, $k^h \geq k^a$.
12. It is assumed that the studied population is a North Island stock and thus the base model replicates the current regulation of this fishery. This focus on the North Island follows the greater availability of data, relative to the South Island, and the increased relevance of findings given that this fishery has yet to be placed under quota control. In line with current management in the North Island, the base model involves no harvest limits, a minimum weight restriction (k^h) consistent with the first adult age class (k^a) (see the previous paragraph), and no maximum weight limit. Since the spawning cohort (k^s) is never exploited, it follows that the eldest harvestable age class (k^H) is k^{s-1} in the base model.
13. This model follows Jellyman et al. (2000) and Hoyle and Jellyman (2002) and assumes that stock size is known and thus exploitation rates can be expressed as a proportion of the adult population. This is a strong assumption, but permits an analysis of optimal harvest.
14. Following the bioeconomic convention of specifying fisheries as capital stocks (Clark 1985), this analysis assumes that a sole manager who aims to maximise Net Present Value (NPV) owns the longfin eel resource. This approach

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[10] This approximate relationship is verified through the comparison of a high number of length-weight relationships reported for longfin eels in Chisnall and Hicks (1993), Beentjes and Chisnall (1997), Beentjes (1999), and Chisnall and Kemp (2000).

considers important economic factors that are absent in biological models of the longfin fishery (Jellyman et al. 2000; Hoyle and Jellyman 2002).

5.4 The economic model

If a positive profit can be earned from the exploitation of all harvestable cohorts ($k=[k^h,...,k^H]$), Conrad (1982) identified that the net revenue ($\pi_{k,t}$) for the *kth* cohort at time t is given by the equation:

$$\pi_{k,t} = P_k \cdot Y_{k,t} - C(X_{k,t}, Y_{k,t}) \tag{5.1}$$

where P_k is the scalar price (measured in dollars) for a tonne of harvested eels from cohort k, $Y_{k,t}$ is the harvest of cohort k at time t, and $C(X_{k,t}, Y_{k,t})$ is the cost function describing the relationship between the cost (measured in dollars) of cropping cohort k at time t as a function of the levels of stock ($X_{k,t}$) and harvest ($Y_{k,t}$) corresponding to this year class. Here it is assumed that $C_Y > 0$ and $C_X < 0$. Price increases with size in reality, but following the nature of this model it is represented through a price that increases with age.

The aggregate net revenue at time t (π_t) is subsequently found by summing that for each harvested cohort:

$$\pi_t = \sum_{k=k^h}^{k^H} \left(P_k \cdot Y_{k,t} - C_{k,t}(X_{k,t}, Y_{k,t}) \right) \tag{5.2}$$

This problem is assumed to involve an infinite horizon, seeing that no obvious terminal condition or period exists. Therefore the total discounted net revenue (π) accruing to the harvest of the studied eel stock is:

$$\pi = \sum_{t=0}^{\infty} \rho^t \cdot \left(\sum_{k=k^h}^{k^H} \left(P_k \cdot Y_{k,t} - C_{k,t}(X_{k,t}, Y_{k,t}) \right) \right) \tag{5.3}$$

where ρ is the discount factor and is defined as $\rho = 1/(1+r)$, where r is the annual discount rate.

5.5 The biological model

The net revenue function presented in equation (5.3) is maximised according to constraints describing the temporal change in eel stocks occurring through both natural effects (that is, growth, migration, recruitment, and natural mortality) and harvest. In a given year *t*, the elver stock (X_0) will depend on the spawning stock (X_{k^s}) of ω years before. This is expressed through the following delay-difference relationship:

$$X_{0,t+\omega} = R(X_{k^s,t}) \tag{5.4}$$

where $X_{0,t+\omega}$ is the elver stock at $t+\omega$ years and $R(X_{k^s,t})$ is a function describing the relationship between the escapement of the spawning cohort in year *t* and the biological recruitment of elvers ω years later. The "recruitment" constraint, describing the entry of juvenile animals into adult cohorts, can be formulated from this equation. As juveniles are not harvested and are subject to density-independent survival, this constraint is:

$$X_{k^a,t+k^a+\omega} = R(X_{k^s,t}) \cdot e^{\sum_{k=0}^{k^a}(G_k - M)} \tag{5.5}$$

where $X_{k^a,t+k^a+\omega}$ is the youngest adult cohort at time $t+k^a+\omega$ and $e^{\sum_{k=0}^{k^a}(G_k - M)}$ is a term describing the total change of biomass through growth and natural mortality between birth and maturity into the adult class. This term consists of two instantaneous rates, physical growth (G_k) and natural mortality (M). G_k represents the growth rate of cohort *k* for juveniles and *M* represents a constant rate of natural mortality.

Additionally, the "growth" constraint describes the way in which the weight of adult cohorts change over time through physical growth, natural mortality, the migration of males at maturity, and harvest. This constraint holds for all adult cohorts except the last; that is, for $k=[k^a,\ldots,k^{s-1}]$. The "growth" constraint is:

$$X_{k+1,t+k^a+\omega+1} = X_{k,t+k^a+\omega} \cdot e^{G(X_{AD,t+k^a+\omega})-M-\mu(k^m)} - Y_{k,t+k^a+\omega} \tag{5.6}$$

where $e^{G(X_{AD,t+k^a+\omega})}$ is a term representing growth for adults in period $t+k^a+\omega$ as a function of the entire adult stock ($X_{AD,t}= \sum_{k=k^a}^{k^s} X_{k,t}$)(where $G_{X_{AD}} < 0$), M is a constant rate of natural mortality, and $\mu(k^m)$ is a term representing the loss of males to migration over the range of ages at which they migrate (k^m). Growth, natural mortality, and the migration of males are assumed to occur before harvest, although these could take place after exploitation without disturbing the qualitative nature of this framework (Bjorndal 1988).

The model takes the form:

Maximise:

$$\pi = \sum_{t=0}^{\infty} \rho^t \cdot \left(\sum_{k=k^h}^{k^H} \left(P_k \cdot Y_{k,t} - C(X_{k,t}, Y_{k,t}) \right) \right) \tag{5.7}$$

Subject to:

$$X_{k^a,t+k^a+\omega} = R(X_{k^s,t}) \cdot e^{\sum_{k=0}^{k^a}(G_k - M)} \tag{5.8}$$

and

$$X_{k+1,t+k^a+\omega+1} = X_{k,t+k^a+\omega} \cdot e^{G(X_{AD,t+k^a+\omega}) - M - \mu(k^m)} - Y_{k,t+k^a+\omega} \tag{5.9}$$

for $k=[k^a,\ldots,k^{s-1}]$.

The modifications necessary to explore alternative management strategies within this model are presented in Section 6.8. These are not required here as the partial analytical solution of the model that follows (Section 5.6) is formulated solely for the base case, as this is deemed representative.

5.6 Analytical solution of the model

Insight into the optimal strategy for managing the longfin fishery can be gained through analytical methods by formulating a Lagrangian system consisting of the model described above in equations (5.7) to (5.9). The Lagrangian is:

$$L=\sum_{t=0}^{\infty}\rho^{t}\cdot\left[\sum_{k=k^{h}}^{k^{H}}(P_{k}\cdot Y_{k,t}-C(X_{k,t},Y_{k,t}))+\rho^{k^{a}+\omega}\cdot\lambda_{k^{a},t+k^{a}+\omega}\cdot(R(X_{k^{s},t})\cdot e^{\sum_{k=0}^{k^{a}}(G_{k}-M)}-X_{k^{a},t+k^{a}+\omega})+\right.$$

$$\left.\sum_{k=k^{a}}^{k^{s-1}}\rho^{k^{a}+\omega+1}\cdot\lambda_{k+1,t+k^{a}+\omega+1}\cdot(X_{k,t+k^{a}+\omega}\cdot e^{G(X_{k,t+k^{a}+\omega})-M-\mu(k^{m})}-Y_{k,t+k^{a}+\omega}-X_{k+1,t+k^{a}+\omega+1})\right] \tag{5.10}$$

This expression requires some explanation. Since the term within the square brackets in the Lagrangian function concerns the net benefits accruing to the fishery in time *t*, each constraint must be discounted to this period. It therefore follows that the marginal values of $X_{k^{a},t+k^{a}+\omega}$ ($\lambda_{k^{a},t+k^{a}+\omega}$) and $X_{k+1,t+k^{a}+\omega+1}$ ($\lambda_{k+1,t+k^{a}+\omega+1}$) are discounted by $\rho^{k^{a}+\omega}$ and $\rho^{k^{a}+\omega+1}$ respectively. The summation term in the second discounted constraint ($\sum_{k=k^{a}}^{k^{s-1}}$) is necessary for the entry of equation (5.9) into the Lagrangian (Conrad 1982; Conrad and Clark 1987), given that in each period the equation of motion will hold for all adult age classes except the last. All of these terms together are discounted by ρ^{t} and summed over time to yield the Net Present Value (NPV) of the fishery. For simplicity, the density-dependent growth term ($e^{G(X_{k,t+k^{a}+\omega})}$) is specified as a function of $X_{k,t+k^{a}+\omega}$, rather than $X_{AD,t+k^{a}+\omega}$, given that $e^{G(X_{AD,t+k^{a}+\omega})}=e^{G(\sum_{k=k^{a}}^{k^{s}}X_{k,t+k^{a}+\omega})}=e^{G(X_{k,t+k^{a}+\omega})}$.

If the cost function is convex, then the necessary conditions for a maximum in discrete-time are (Conrad and Clark 1987; Conrad 1999):

$$\frac{\partial L}{\partial Y_{k,t}}=0 \tag{5.11}$$

for $k=[k^{h},\ldots,k^{H}]$.

$$\frac{\partial L}{\partial X_{k,t}} = 0 \tag{5.12}$$

for $k=[k^a,\ldots,k^s]$.

$$\frac{\partial L}{\partial(\rho^{k^a+\omega} \cdot \lambda_{k^a,t+k^a+\omega})} = 0 \tag{5.13}$$

$$\frac{\partial L}{\partial(\rho^{k^a+\omega+1} \cdot \lambda_{k+1,t+k^a+\omega+1})} = 0 \tag{5.14}$$

for $k=[k^a,\ldots,k^{s-1}]$.

In a multiple-cohort population in which year classes can be divided according to reproductive capability (that is, spawning/non-spawning) then equation (5.12) must be calculated twice, once for each partition. Since only females are included within the spawner-recruitment relationship and this year class (k^s) is not exploited, equation (5.12) is calculated once for each harvestable cohort ($k=[k^h,\ldots,k^H]$, which is analogous to $k=[k^a,\ldots,k^{s-1}]$ in the base model) and once for the spawning cohort ($k=k^s$).

The first First Order Condition (FOC) concerns the derivative of the Lagrangian with respect to harvest of cohort k at time t ($Y_{k,t}$):

$$\frac{\partial L}{\partial Y_{k,t}} = \rho^t \cdot \left[P_k - \frac{\partial C(X_{k,t}, Y_{k,t})}{\partial Y_{k,t}} - \rho \cdot \lambda_{k+1,t+1} \right] = 0 \tag{5.15}$$

for $k=[k^h,\ldots,k^H]$.

While the derivative with respect to the profit function is straightforward, the origin of the term $\rho \cdot \lambda_{k+1,t+1}$ warrants some discussion. Given that the derivative is taken with respect to $Y_{k,t}$, and the term $Y_{k,t+k^a+\omega}$ is present in the Lagrangian above (equation (5.10)), it is necessary to regress $k^a+\omega$ periods for this term to become $Y_{k,t}$. It follows that its factors become $\rho \cdot \lambda_{k+1,t+1}$. The $Y_{k,t}$ term disappears during differentiation through the implementation of the power-function rule (Chiang 1984) and thus the $\rho \cdot \lambda_{k+1,t+1}$ term remains. After simplification the first FOC becomes:

$$P_k = \frac{\partial C(X_{k,t}, Y_{k,t})}{\partial Y_{k,t}} + \rho \cdot \lambda_{k+1,t+1} \tag{5.16}$$

for $k=[k^h,...,k^H]$.

The second FOC is the derivative of the Lagrangian with respect to $X_{k,t}$ for all harvested cohorts. This is:

$$\frac{\partial L}{\partial X_{k,t}} = \rho^t \cdot \left[-\frac{\partial C(X_{k,t}, Y_{k,t})}{\partial X_{k,t}} + \rho \cdot \lambda_{k+1,t+1} \cdot e^{G(X_{k,t})-M-\mu(k^m)} (1 + X_{k,t} \cdot \frac{\partial G(X_{k,t})}{\partial X_{k,t}}) \right] - \rho^t \cdot \lambda_{k,t} = 0 \tag{5.17}$$

for $k=[k^a,...,k^{s-1}]$.

While the derivative with respect to the profit function is straightforward, once again it is necessary to regress a number of periods in order to identify an $X_{k,t}$ term. The first term containing $X_{k,t}$ in the growth constraint (equation (5.9)) present in the Lagrangian function is $X_{k^a,t+k^a+\omega}$, which is multiplied by $e^{G(X_{k,t+k^a+\omega})-M-\mu(k^m)}$. In order for an $X_{k,t}$ term to be present, it is therefore necessary to back up $k^a+\omega$ periods. It follows that the terms that it is multiplied by become $\rho \cdot \lambda_{k+1,t+1}$. This term ($\rho \cdot \lambda_{k+1,t+1} \cdot X_{k,t} \cdot e^{G(X_{k,t})-M-\mu(k^m)}$) has its derivative identified using the generalised exponential function rule and the product rule (Chiang 1984).

The last term on the LHS of equation (5.17) arises from a similar situation to that described above, except here the $X_{k,t}$ term, corresponding to the last term in the Lagrangian ($X_{k,t+k^a+\omega+1}$), is found after regressing $\omega+k^a+1$ years. The $X_{k,t}$ term disappears during differentiation through the implementation of the standard power-function rule (Chiang 1984), and thus only the discounted Lagrangian multiplier associated with this term remains. Moving this expression to the RHS and simplifying, the second FOC is obtained:

$$-\frac{\partial C(X_{k,t}, Y_{k,t})}{\partial X_{k,t}} + \rho \cdot \lambda_{k+1,t+1} \cdot e^{G(X_{k,t})-M-\mu(k^m)} \left[1 + X_{k,t} \cdot \frac{\partial G(X_{k,t})}{\partial X_{k,t}} \right] = \lambda_{k,t} \tag{5.18}$$

for $k=[k^a,\ldots,k^{s-1}]$.

The third FOC is the derivative of the Lagrangian with respect to $X_{k,t}$ for the spawning cohort, $X_{k^s,t}$. This is:

$$\frac{\partial L}{\partial X_{k,t}}=\rho^{k^a+\omega}\cdot\lambda_{k^a,t+k^a+\omega}\cdot\frac{\partial R(X_{k^s,t})}{\partial X_{k,t}}\cdot e^{\sum_{k=0}^{k^a}(G_k-M)}=0 \tag{5.19}$$

for $k=k^s$.

No derivative exists for either the objective function (equation (5.7)) or growth constraint (equation (5.9)) as the spawning cohort is not cropped and does not grow within the model. No backing up is required, in contrast to the other FOCs, given that $X_{k^s,t}$ is in the current period t.

The last two FOCs are merely restatements of the recruitment and growth constraint respectively, as expected when taking the partial derivative of a Lagrangian function with respect to a discounted multiplier. The first is:

$$\frac{\partial L}{\partial(\rho^{k^a+\omega}\cdot\lambda_{k^a,t+k^a+\omega})}=R(X_{k^s,t})\cdot e^{\sum_{k=0}^{k^a}(G_k-M)}-X_{k^a,t+k^a+\omega}=0 \tag{5.20}$$

which simplifies to the recruitment constraint:

$$X_{k^a,t+k^a+\omega}=R(X_{k^s,t})\cdot e^{\sum_{k=0}^{k^a}(G_k-M)} \tag{5.21}$$

The second is:

$$\frac{\partial L}{\partial(\rho^{k^a+\omega+1}\cdot\lambda_{k+1,t+k^a+\omega+1})}=X_{k,t+k^a+\omega}\cdot e^{G(X_{k,t+k^a+\omega})-M-\mu(k^m)}-Y_{k,t+k^a+\omega}-X_{k+1,t+k^a+\omega+1}=0 \tag{5.22}$$

for $k=[k^a,\ldots,k^{s-1}]$.

Which simplifies to the growth constraint:

$$X_{k+1,t+k^a+\omega+1} = X_{k,t+k^a+\omega} \cdot e^{G(X_{k,t+k^a+\omega})-M-\mu(k^m)} - Y_{k,t+k^a+\omega} \tag{5.23}$$

for $k=[k^a,...,k^{s-1}]$.

This yields the FOCs collectively as:

$$P_k = \frac{\partial C(X_{k,t}, Y_{k,t})}{\partial Y_{k,t}} + \rho \cdot \lambda_{k+1,t+1} \tag{5.24}$$

for $k=[k^h,...,k^H]$.

$$-\frac{\partial C(X_{k,t}, Y_{k,t})}{\partial X_{k,t}} + \rho \cdot \lambda_{k+1,t+1} \cdot e^{G(X_{k,t})-M-\mu(k^m)} \left[1 + X_{k,t} \cdot \frac{\partial G(X_{k,t})}{\partial X_{k,t}}\right] = \lambda_{k,t} \tag{5.25}$$

for $k=[k^a,...,k^{s-1}]$.

$$\rho^{k^a+\omega} \cdot \lambda_{k^a,t+k^a+\omega} \cdot \frac{\partial R(X_{k^s,t})}{\partial X_{k,t}} \cdot e^{\sum_{k=0}^{k^a}(G_k-M)} = 0 \tag{5.26}$$

for $k=k^s$.

$$X_{k^a,t+k^a+\omega} = R(X_{k^s,t}) \cdot e^{\sum_{k=0}^{k^a}(G_k-M)} \tag{5.27}$$

$$X_{k+1,t+k^a+\omega+1} = X_{k,t+k^a+\omega} \cdot e^{G(X_{k,t+k^a+\omega})-M-\mu(k^m)} - Y_{k,t+k^a+\omega} \tag{5.28}$$

for $k=[k^a,...,k^{s-1}]$.

The first FOC (equation (5.24)) is identical to that identified by Conrad (1982) for his generalised multiple-cohort framework and is the multi-dimensional analogue of the expression for the standard harvesting problem in discrete-time (for example that derived by Conrad 1999, p12). This equation requires that under the optimal management policy the net benefit accruing to the marginal harvested unit of cohort k (P_k) be equated to the marginal cost of harvest ($\frac{\partial C(X_{k,t}, Y_{k,t})}{\partial Y_{k,t}}$, which is positive by definition) and the opportunity cost of harvest (user cost) ($\rho \cdot \lambda_{k+1,t+1}$). This user cost is the discounted value of an additional tonne of eels in the next cohort, one time period in the future (Conrad 1982), and therefore represents the future cost associated with current harvest.

The second FOC (equation (5.25)) specifies how all harvested cohorts should be managed under the optimal strategy and resembles the FOC for the stock-related derivative within the standard harvesting problem (see Conrad 1999, p12). The optimal management of a harvested longfin cohort requires that the marginal value of an additional tonne of eels in cohort k at time t ($\lambda_{k,t}$) is equated to the sum of the benefit of decreased costs arising from an additional unit of stock in cohort k at time t ($\frac{\partial C(X_{k,t}, Y_{k,t})}{\partial X_{k,t}}$, which is negative by definition) and the marginal benefit accruing to an unharvested tonne of eels in the next period ($\rho \cdot \lambda_{k+1,t+1} \cdot e^{G(X_{k,t})-M-\mu(k^m)}\left[1+X_{k,t} \cdot \frac{\partial G(X_{k,t})}{\partial X_{k,t}}\right]$) (Conrad 1999). The latter is both a positive and negative function of the stock level, as while an unharvested unit of biomass is available for harvest in the next period, leaving this stock also decreases growth as competition for food and space increases. These effects are observable in that this expression ($\rho \cdot \lambda_{k+1,t+1} \cdot e^{G(X_{k,t})-M-\mu(k^m)}\left[1+X_{k,t} \cdot \frac{\partial G(X_{k,t})}{\partial X_{k,t}}\right]$) will increase with increases in $X_{k,t}$ and decrease with the $e^{G(X_{k,t})-M-\mu(k^m)}$ and $\frac{\partial G(X_{k,t})}{\partial X_{k,t}}$ terms, which are both inversely related to $X_{k,t}$ by definition (given density-dependent growth). This effect aids the convergence of the numerical model to steady-state because it promotes the stabilisation of cohort stocks.

Equation (5.25) can also be solved for the optimal (harvested) cohort stock level ($X_{k,t}$* for $k=[k^h,...,k^H]$). Adding $\frac{\partial C(X_{k,t}, Y_{k,t})}{\partial X_{k,t}}$ to both sides, dividing both sides by

$\rho \cdot \lambda_{k+1,t+1} \cdot e^{G(X_{k,t})-M-\mu(k^m)}$, subtracting unity from both sides, and then dividing both sides by $\frac{\partial G(X_{k,t})}{\partial X_{k,t}}$ provides:

$$X_{k,t}^* = \frac{\left[\frac{\lambda_{k,t} + \frac{\partial C(X_{k,t}, Y_{k,t})}{\partial X_{k,t}}}{\rho \cdot \lambda_{k+1,t+1} \cdot e^{G(X_{k,t})-M-\mu(k^m)}} \right] - 1}{\frac{\partial G(X_{k,t})}{\partial X_{k,t}}} \tag{5.29}$$

The way in which cohort growth is affected by the stock level ($\frac{\partial G(X_{k,t})}{\partial X_{k,t}}$) dominates this expression, demonstrating the important role that density-dependent growth plays in the determination of the biomass of harvestable year classes and therefore cropping.

The third FOC (equation (5.26)) also differs from that of Conrad (1982), given that this analysis involves a semelparous species, a density-dependent spawner-recruitment relationship, and a reproductive cohort that is not harvested. This expression specifies that for the optimal management of the spawning cohort (k^s), the discounted value of the young reaching the first adult cohort, as a function of the spawning stock $t+\omega$ years before, should equal zero. This is intuitive given that there are benefits accruing to a higher/lower spawning biomass if the discounted marginal value of the first adult cohort is positive/negative. It follows that the optimal level is reached where the discounted value of the first adult cohort is zero.

Equations (5.27) and (5.28) are simply the recruitment and growth constraints as specified earlier.

5.7 Conclusions

The partial solution of the analytical model demonstrates a number of factors that are important to the determination of the optimal management strategy. User cost plays a key role, as harvest in the current period will have significant ramifications for future cropping due to the long lives of longfins. First, exploitation of a younger year class prevents its harvest in later periods when it would be larger due to growth. Second, since price is increasing in age, the exploitation of biomass in a younger year class also prevents the harvest of these eels in later years when a given unit of biomass is more valuable *ceteris paribus*. Third, exploitation of a younger year class will decrease the future ability of this biomass to lower harvesting costs (through the inclusion of a stock-related cost externality). Fourth, harvest denies individuals the chance to make a reproductive contribution, given that all fishing mortality occurs before spawning. Additionally, cropping has a user benefit, in that exploitation will increase growth due to decreased competition among individuals. The partial solution of this model also indicates that it is necessary to balance the effect of diminishing returns to an increasing spawning population.

While this derivation provides some insight into the characteristics of a solution, the complexity of this model prevents further analytical investigation to the level of detail required. The remainder of this analysis therefore relies on the numerical solution of the developed model.

6. THE NUMERICAL MODEL

6.1 Introduction

The analytical complexity of the model developed in Chapter 5 identifies the need to use numerical methods to research the optimal management of the longfin eel. This involves a significant amount of estimation and data collection because of a lack of biological and economic information for this species.

In this chapter, the characteristics of the study region and the life cycle of its longfin stock are described. The estimation of the parameters and relationships included within this model are then outlined. The numerical model and its solution are then discussed.

6.2 The study region

Longfins cannot be modelled as a national stock because of a lack of biomass information and heterogeneous population density and growth (Annala et al. 2001). To portray recruitment overfishing and the recovery of this fishery, it is necessary to study a region in which environmental differences are minimised, but which is large enough to suggest a relationship between this area's stock and recruitment into it. The availability of sufficient information is also important.

The most suitable area is the lower section of the Waikato River (37°33'S, 175°09'E). The Waikato is New Zealand's longest river and travels a distance of 425 kilometres from Lake Taupo to its entry into the Tasman Sea at Port Waikato. There are currently eight hydroelectric dams on this river, each of which is designed so that water flows directly from one station to the next. The Karapiro dam is the closest to the sea and is around 150km from the river's mouth. This dam has limited both the in- and out-migration of longfins since its construction in 1947-48. This virtual "closure" of the area between the Karapiro dam and the sea at Port Waikato (henceforth referred to as the lower Waikato River) defines the boundaries of what may be regarded, for the means of this analysis, as a single stock. The study area is presented in Figure 6.1, and involves the waterway flowing from the Karapiro Dam to Ngaruawahia and from here to Port Waikato. Green areas denote elevation in Figure 6.1, with darker regions representing lower areas.

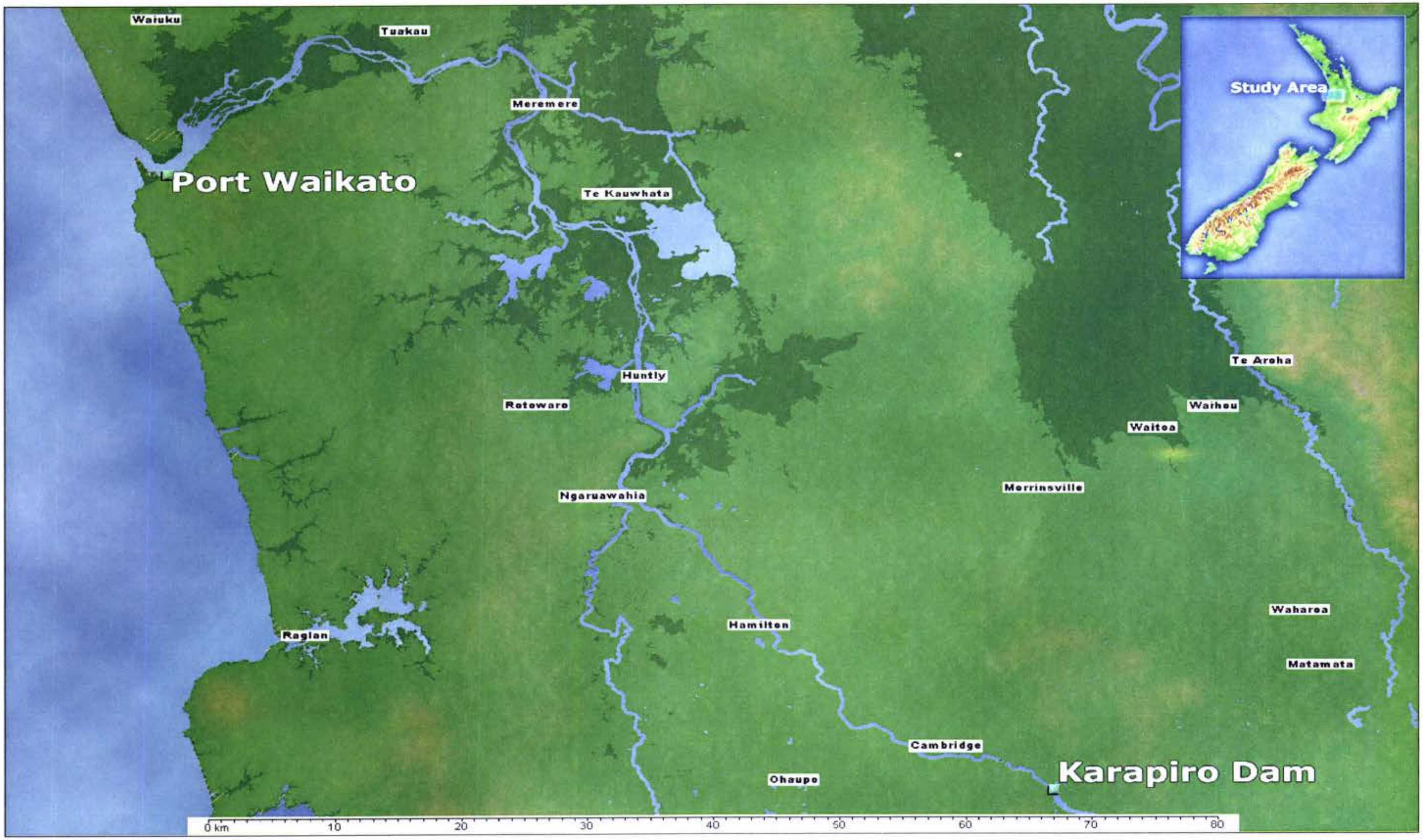

Figure 6.1. Location of the study area.

The large size of this area and the favourable growth environment for eels here, relative to that in other locations (Chisnall 1989; Chisnall and Hicks 1993; Chisnall et al. 2002), suggest that some relationship exists between spawners leaving this system and the subsequent recruitment of elvers. This validates the specification of a spawner-recruitment relationship, given the need to incorporate this to portray the severe recruitment overfishing present in longfin populations.

This area has played an important role in a region that has historically supported the largest eel fishery in New Zealand (Chisnall 1989). CPUE data is of too poor quality to identify trends in exploitation (Jellyman 1993), but from historical data it can be inferred that the high yields of earlier years are now gone. For example, annual harvest in this region has fallen by around 70 percent over the last twenty years. This increases the relevance of this analysis as it specifically studies the recovery and management of longfins in a region that once supported strong freshwater eel populations (Chisnall 1994) and high levels of harvest (Todd 1981b; Chisnall 1989).

The longfin eel stocks of the Waikato River and various tributaries have also been well studied (Chisnall 1989; Chisnall and Hicks 1993; Chisnall 1994; Hicks and McCaughan 1997; Chisnall et al. 2002). The provision of biomass density data is particularly important (Chisnall and Hicks 1993; Hicks and McCaughan 1997; Chisnall et al. 2002) since this is necessary for estimating initial conditions and density-dependent relationships.

Chisnall et al. (2002) provides the most detailed study of longfin density and growth in a consistent environment (the Ahirau stream). Significant reliance on these findings is valid for a number of reasons. First, the growth of a virgin (that is, unfished) population in the Ahirau stream is likely to resemble that of one in the lower Waikato River, given that the Ahirau is a tributary of this river and also runs through pastoral land (Chisnall and Hicks 1993; Hicks and McCaughan 1997; Chisnall et al. 2002). Second, it is the most studied waterway in close proximity of the lower Waikato River. This permits the use of growth relationships and biomass estimates from a uniform environment.

6.2.1 Growth within the study region

It is assumed that animals share a common growth relationship over this long length of river. This seems a fair assumption, because although significant individual variability would exist (Jellyman 1997; Francis and Jellyman 1999; Hoyle and Jellyman 2002), this river passes primarily through pastoral land and thus eels are likely to share a similar growth relationship along its course (Chisnall and Hicks 1993).

Individuals within each cohort are described by a mean weight and length (Section 5.3). The weight (1.46g) and length (103.4mm) of longfin elvers (k=0) are the means of samples taken from a population of 1.2 million shortfin and longfin specimens caught at the Karapiro dam between 23 December 1995 and 20 March 1996 (Beentjes et al. 1997). To minimise error due to extrapolation from the growth relationship used for older eels, estimates of weight and length for the second year class (k=1) are identified by finding the midpoint between those for the youngest year class (k=0) and the third cohort (k=2). The weight and length for the third year class (k=2) and all of those older (k=[3,…,31]) are calculated from growth relationships presented in Chisnall et al. (2002).

Weight-at-age figures are identified using an allometric specification in which a length-at-age relationship is substituted. This provides for the calculation of a density-dependent growth function for adult cohorts (Section 6.6).

Length-at-age is identified from the linear relationship between age (k) (in years) and length (L) (in millimetres) reported by Chisnall et al. (2002):

$$L = 58.7 + 35.8 \cdot k \qquad (6.1)$$

The R^2 statistic (R^2=.81) provides some confidence in the accuracy of this relationship, given the range of the size (231-1095mm) and age (four to twenty-five years) of animals involved. Although the low number of studied individuals (n=29) remains of some concern, the use of a relationship consistent with the adopted density data is deemed more important. This approach is supported by the small difference observed between the relationship in equation (6.1) and that of other populations in similar habitats (Chisnall and Hicks 1993; Hicks and McCaughan 1997; Beentjes and Chisnall 1998).

The allometric relationship between length (L) (in millimetres) and weight (W) (in grams) reported by Chisnall et al. (2002) is described by the following equation:

$$W = .00000046 \cdot L^{3.28} \tag{6.2}$$

The identification of this relationship involved a sample of 132 animals within a range of 8.3g to 4191g. A strong association was identified, with an R^2 statistic of .98.

The mean length and weight of each year class are presented in Table A1.1 in Appendix 1.

6.2.2 Sexual maturity within the study region

Longfin populations exhibit wide variation in gender composition, with females dominating inland habitats and males dominating areas closer to the coast (Jellyman et al. 2000). In order to present a model that is representative of the longfin fishery as a whole, this analysis follows the studies of Francis and Jellyman (1999) and Jellyman et al. (2000) in assuming equal sex ratios for recruiting eels. This seems a fair assumption because of the size of the study area.

All females are assumed to mature at the age at which length-at-age is closest to that corresponding to the mean size of migrant female longfins (1156mm) (Todd 1980). The age at which length-at-age is closest to this estimate is 31 years (see Table A1.1 in Appendix 1).

In contrast, males migrate over a range of ages in this model. The proportion of males migrating at each length is calculated from the Makara Stream (n=362) sample of Todd (1980). The appropriate proportions of migrants for each age class are then identified using the length-age relationship identified by Chisnall et al. (2002) (equation (6.1)). These proportions are then multiplied by one half, in light of the equal division of the population between male and female individuals. To convert these proportions into instantaneous rates of male migration (μ), the following formula is then used:

$$\mu = \ln[1 - prop_m(k)] \tag{6.3}$$

where $prop_m(k)$ is the proportion of the total stock that is lost to the annual migration of males in each affected age group k. The results of this calculation provide an estimate of the proportion of the stock that is lost annually to male migration for each year class concerned (Table 6.1). Strong sexual dimorphism is evident as the age of males at sexual maturity (Table 6.1) is well below the mean age of migrating females in this environment, which is around 31 years.

Table 6.1. The instantaneous rate of male migration (μ) for each affected age class.

Age (*k*)	μ
12	-0.015
13	-0.035
14	-0.105
15	-0.114
16	-0.149
17	-0.057
18	-0.025

6.2.3 Life cycle within the study region

A diagram of the longfin life cycle within the stock described by the model is provided in Figure 6.2. This life cycle represents a number of simplifying assumptions. First, females are assumed to take six months to swim to oceanic spawning grounds from their freshwater habitat with all cohort members reaching maturity at the same age. Second, it is assumed that young take eighteen months from the time they hatch in the ocean until they arrive in their freshwater home range. Therefore, the lag (ω) between escapement and the resulting biological recruitment of offspring (elvers or k=0 within the model) is two years. Third, using the growth relationships mentioned above, eels are assumed to reach the existing minimum harvestable weight at eleven years of age, as the mean weight of individuals within this cohort is the closest to this weight (220g) (see Table A1.1 in Appendix 1). Although this may be manipulated to portray the effects of different weight restrictions, the division between juvenile and adult classes is always assumed to exist between eels of zero and ten years of age and eels of eleven years and above. This division arises from dimorphism in dietary preference, given that eels above 45cm are predominantly piscivorous. See Section 5.3 for further details.

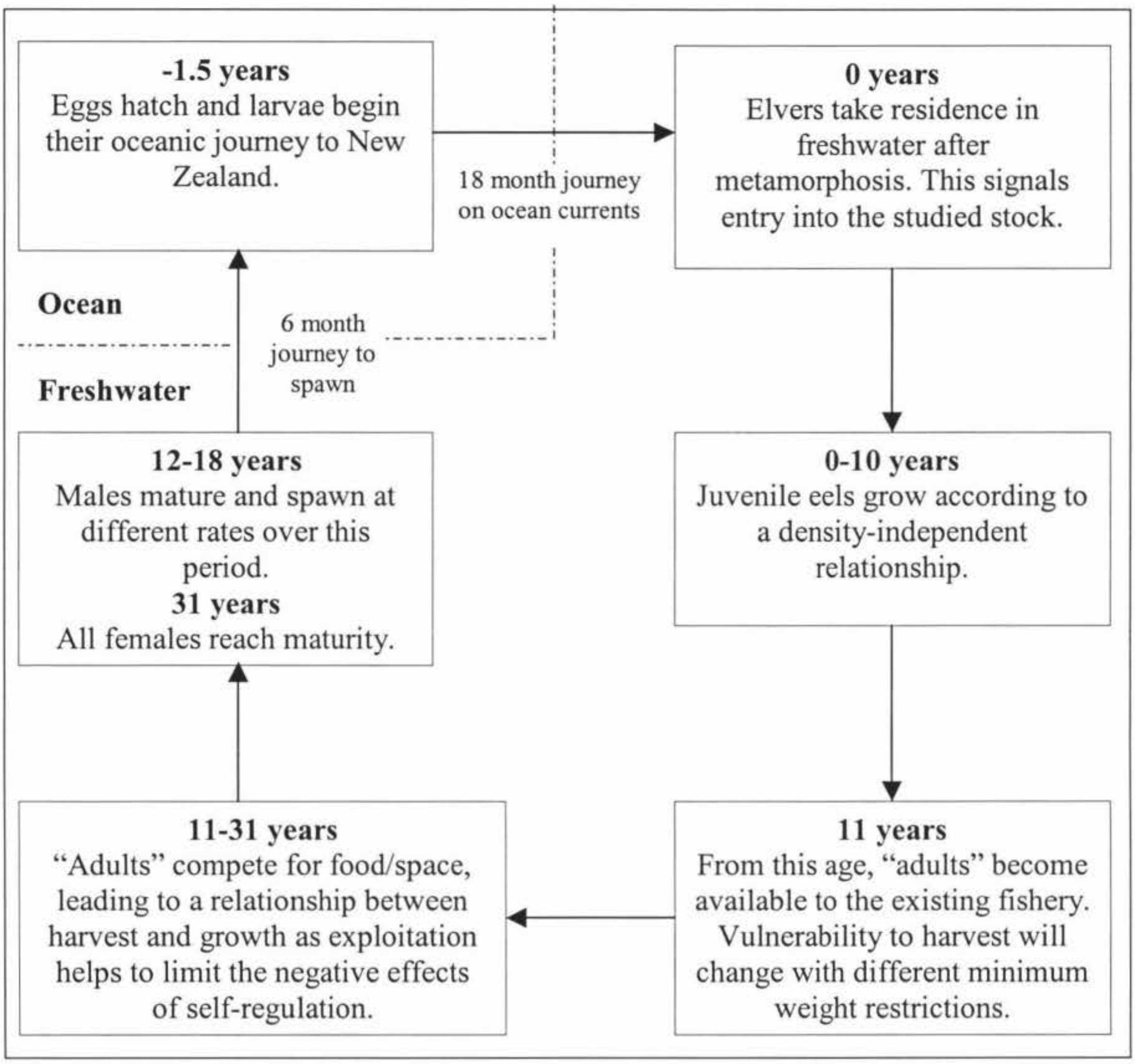

Figure 6.2. Generalised life cycle of the lower Waikato River eel stock.

The life cycle depicted here (Figure 6.2) depends strongly on the growth of the studied stock. Nonetheless, the adopted growth relationship provides for general insight because the mean age of longfin eels first reaching harvestable size in the North Island is nine years (Annala et al. 2001), compared to eleven years in this analysis.

This study follows Conrad (1982) and assumes that the ages that define the sexual maturity of individuals do not change. This approach represents a compromise between the contrasting views of the determinants of sexual maturity for the longfin (see Section 2.3.2 and Section 5.3). This provides a pessimistic view of recovery if length is the primary determinant of sexual maturity, because low initial stock densities would promote rapid growth within this model (given the portrayal of density-dependent growth) and thus animals would spawn at younger ages. This approach is preferable to one that portrays more optimism as it provides a conservative estimate of outcomes in an analysis of management strategies for a fishery in which significant uncertainty exists. This is consistent with the need for a precautionary approach to management identified by Hoyle and Jellyman (2002).

6.3 Formulation of estimates for the objective function

6.3.1 Estimation of prices received for harvested eels

Prices are assumed constant. This seems a fair assumption for New Zealand's eel products as a wide range of close substitutes are present in the global market and New Zealand constitutes only a small proportion of world supply. For example, the on growing of wild eels in eastern Asia produces around 150,000 tonnes of eel product annually (Moriarty and Dekker 1997), in contrast to a mean annual harvest of 1457 tonnes in New Zealand since 1980. Additionally, the improved quality of New Zealand's eels, due to increased fat and oil content arising from decreased population densities after extensive exploitation (McDowall 1990), now puts them in direct competition with the majority of these foreign products.

The representation of a price that increases with size/age is problematic in a model incorporating density-dependent growth. For example, longfins would reach the most valuable price range more quickly at low stock levels, while growth would be retarded at high densities and thus eels would take longer to reach the highest weight bracket. This analysis follows Conrad (1982) and assumes that the ages that define different price brackets are constant. This provides for mathematical tractability and is validated in Chapter 7, where the robustness of the optimal harvest profile to constant prices across all cohorts is demonstrated.

Prices are estimated from telephone interviews with four of New Zealand's five major eel processors, who wished for their individual price information to remain confidential. These prices are for all eels transported to the processing plant by processors after harvest, which is the primary method of reception (M. Kuijten, *pers.*

comm., 1 May 2002). Therefore these price figures include the cost of transporting harvested eels to processing. The loss of eels to injuries sustained during harvesting and handling is ignored, as anecdotal evidence suggests that these losses are very small (M. Kuijten, *pers. comm.*, 1 May 2002).

Harvesters are paid different rates according to the size of individual animals, with animals within higher weight brackets receiving a higher price. The prices used are the mean price for each range of year classes that correspond to these weight brackets. The relationship between weight and age is determined from the mean weight of individuals in each cohort (Table A1.1 in Appendix 1). The price for each bracket of harvestable cohorts is presented in Table 6.2. All monetary values in this model are specified in New Zealand dollars.

Table 6.2. The mean scalar price per tonne (P_k) for each bracket of harvestable year classes.

Weight bracket	Age bracket (years)	Price received (dollars/tonne)
220g-499g	11-14	4350
500g-999g	15-17	4540
1kg-1.49kg	18-20	4850
1.5kg +	21-30	5950

Table 6.2 demonstrates that price increases with longfin age, but lifts most significantly when age is above 21 years in the model (the equivalent of 1.5kg). Price is 27 percent higher than that paid for the lowest weight bracket ($4350/tonne) at this point. It is important to note here that animals in the larger size bracket will always be female, both in reality (Todd 1980) and in the model, as in the latter no males live past 18 years of age (Table 6.1). This demonstrates the contribution of economic incentives to the overexploitation of this slower maturing gender.

6.3.2 Formulation of a cost function for the eel fishery

The estimation of a production function for the longfin eel fishery is problematic due to a lack of stock information. Additionally, insufficient data prevents the use of Virtual Population Analysis (VPA) or Relative Abundance Analysis (RAA) to estimate population size (Quinn and Deriso 1999). It is also not possible to use the former on a life stage basis, as done for European eels (*A. anguilla*) (Dekker 2000), because longfins are only removed at one life stage (adulthood). The use of catch and effort data to estimate stock numbers (Ricker 1975) is also unsuitable given a lack of catch-effort data for the specific stock that is analysed, the low quantity and poor quality of regional series, and the collinearity associated with using these estimates in the estimation of a production function. Additionally, the use of a proxy based on mean harvest can complicate estimation through endogeneity (Alvarez 2002). Simulation from recruitment data is also problematic given intermittent sampling, short series, and variation in the effort applied to annual collection. Yet even if these problems arising from poor information are overcome, the correct use of exogenous population data in production function estimation has yet to be satisfactorily determined (Schott 2002).

The positive stock-related cost externality is therefore represented through the inclusion of a harvest-stock ratio term in the cost function (Conrad 1999):

$$C_{k,t} = c \cdot Y_{k,t} \cdot \left(\frac{Y_{k,t}}{X_{k,t}^{i}} \right) \tag{6.4}$$

where c is the cost of harvesting one tonne of eels (in dollars per tonne) and i is an exponent describing the strength of diminishing returns to the stock of cohort k at time t. This exponent is set to unity in the base model, but the effects of altering this coefficient will be explored through sensitivity analysis given that inefficiencies, such as gear saturation, may exist in reality.

While this cost function differs from those involving traditional production functions based in production theory, for example the Cobb-Douglas specification (Zellner et al. 1966), it portrays stock-related cost effects without requiring specific population data. A more accurate specification would be appropriate if a suitable method of estimation or better information becomes available.

6.3.3 Estimation of the cost parameter

The cost of harvesting one tonne of eels (c) is estimated through a mail survey of all commercial eel harvesters in the Waikato region. The entire sampling population was surveyed since only twenty harvesters are currently authorised to fish in this region. These harvesters make up around a third of the total number of permit holders in the North Island (Ministry of Fisheries data). The names and addresses of these permit holders were obtained from the Ministry of Fisheries.

The primary motivation for the sole surveying of harvesters who are licensed to fish in the Waikato region is the estimation of a cost parameter of more relevance to the model, given that all participants fish in the general region in which this study is based. While increasing the sample population to include fishers from other areas would have provided for an increased number of replies, the cost estimate would have been less accurate due to variability in fishing efficiency between areas (Jellyman 1993).

With nine completed replies after three weeks, and three replies stating a wish not to participate, the remainder of the harvesters were interviewed over the phone and an additional four replies were subsequently obtained. Thus a total of thirteen replies from a possible twenty were acquired, giving an effective respondence rate of 65 percent.

While the survey sought to identify information that could be used to gain an estimate of the cost of fishing (including depreciation and the opportunity cost of harvesting in terms of alternative employment), it also aimed to gain some understanding of how commercial harvesters view the management of the fishery. The precise objectives of this survey were:

1. to determine the educational background of fishers so as to gain some understanding of the opportunity cost of harvest in terms of alternative employment opportunities available to them
2. to identify the nature and age of equipment used by harvesters
3. to estimate the cost of harvesting one tonne of eels
4. to gain some understanding of what commercial fishers believe the state of the fishery to be and how well they think it is managed, and
5. to gain insight into the opinions of commercial harvesters in regard to future management options.

By satisfying these objectives, this survey allowed for the estimation of an important parameter, alongside the provision of insight into additional areas, such as the educational background of harvesters, which existing research into this fishery does not provide.

Details of the survey are provided in Appendix 2 and a full analysis of results is presented in Appendix 3, but a short summary of cost considerations is provided here. The variable cost portion of the harvesting cost parameter (*c*) is \$610.65/tonne. Following Conrad (1982) and Campbell et al. (1993), depreciation and the opportunity cost of harvest are also included in the cost parameter. However, no annual loss to depreciation is identified from the survey, given the low capital outlay for harvesters (especially in comparison to many ocean fisheries), the significant age of equipment used, and the depreciation rates involved. The educational background of fishers is used to identify the opportunity cost of harvest in terms of foregone income. An estimate of \$1769.49/tonne (2dp) is obtained. Added to the variable cost component provided above (\$610.63/tonne), this gives a total harvesting cost of \$2380.12 per tonne. See Appendix 3 for a detailed discussion of the estimation of these parameters.

6.3.4 Calculation of the initial discount rate

An analysis of past long-term government bond rates is used to calculate an appropriate discount rate. This is appropriate since the problem is cast in infinite time. The mean annual real ten-year secondary market government bond rate for December 1994 to December 2001 is used, which is 5.09 percent. This is rounded to 5 percent. Using a value from this period avoids any bias arising from the higher inflation levels (and therefore interest rates) experienced within New Zealand throughout the 1980s and early 1990s, while also allowing for the low inflation commitment outlined in the 1989 Reserve Bank of New Zealand Act to take effect. The latter increases the validity of the chosen discount rate, as this policy commitment to low and stable inflation provides a legislative foundation for the belief that future rates may remain relatively low.

All parameters within the objective function are assumed to be constant over time, as in many bioeconomic analyses (Campbell et al. 1993; Larkin and Sylvia 1999; Lee et al. 2000). This seems a fair assumption for price because New Zealand's eel products comprise only a small proportion of world supply. This approach is also necessary for the other parameters because of a lack of information.

6.4 The measurement of stock levels

It is necessary to form an estimate of the size of the longfin stock in the lower Waikato River for the estimation of a density-dependent growth term, a stock-recruitment function, and initial conditions for the model. To form this estimate, it was first necessary to calculate the surface area of this region using Geographic Information Systems (GIS). This was done by Matthew Keene (GIS Technician, New Zealand Centre for Precision Agriculture, Massey University, Palmerston North). First, two arcview polygon shapefiles of the Waikato River and all of its tributaries from Karapiro dam to the entry of this river into the Tasman Sea were isolated from the New Zealand topographic map series. These shapefiles were then merged and a new theme placed over them. This theme was then cropped to leave only the Waikato River. An extension from Arcview then calculated the area of this new theme. The total area was 2576.26 hectares.

The longfin density used to identify the initial stock estimate is that for the 1988 year in Chisnall et al. (2002). This figure is pre-harvest and includes all eels of the minimum weight permitted for commercial harvest (220g) and above. The density used is 26.8g/m^2. Given that the total area is 2576.26 hectares and one hectare is equal to 10,000m^2, the total size of the area is 25,762,600m^2. This gives an initial virgin stock estimate for the weight of eels above 220g of 690 tonnes (rounded to nearest whole number). While this method of stock estimation ignores the contribution of tributaries and spatial considerations, the territoriality of longfins and a lack of information provide some encouragement for this technique (Schnute 1987; Hillborn and Mangel 1997).

6.5 Estimation of the instantaneous survival rate for juveniles

The growth of eels less than 45cm in length is classified as density-independent and cannibalism by older year classes is not included due to a lack of information. The instantaneous growth rate (G_k) for juvenile cohort k is identified through the following formula:

$$G_k = \ln\left(\frac{W_{k+1}}{W_k}\right) \tag{6.5}$$

where W_k is the mean weight of an individual fish within year class k and W_{k+1} is the mean weight of an individual in the subsequent year class. The weight-at-age figures for each respective year class (Table A1.1 in Appendix 1) and this equation are used to calculate instantaneous growth rates (G_k) for each discrete growth period for juveniles. These values are then summed (denoted as $\sum_{k=0}^{k^a} G_k$) to represent growth between biological recruitment and entry into the first adult year class (k^a). It is identified that $\sum_{k=0}^{k^a} G_k$ =5.09 (2dp).

This analysis follows Jellyman et al. (2000) and Hoyle and Jellyman (2002) and specifies the instantaneous rate of natural mortality (M) for juveniles as .04 year^{-1}. This is the rounded mean of the rates identified for adult longfins by Francis and Jellyman (1999) (.042 year^{-1} and .036 year^{-1}). This adult rate is used due to a lack of specific information for juveniles. These values are summed (denoted as $\sum_{k=0}^{k^a} -M$) to represent natural mortality between biological recruitment and entry into the first adult cohort. It is identified that $\sum_{k=0}^{k^a} -M = .44$. The expression for the total survival rate ($e^{\sum_{k=0}^{k^a}(G_k - M)}$) governing the growth and natural mortality of eels between biological recruitment and entry into the first adult cohort is therefore $e^{(5.09-.44)}$ or $e^{4.65}$.

6.6 Estimation of the instantaneous density-dependent growth rate for adults

The growth constraint presented in equation (5.9) incorporates natural mortality, the loss of males at sexual maturity, and a density-dependent growth relationship. The level of natural mortality (M) for adults is the rounded mean of those rates listed for longfins by Francis and Jellyman (1999), .04 year^{-1} (see Section 6.5), while the rates used to denote the loss of males to migration ($\mu(k^m)$) are presented in Table 6.1. This section describes the estimation of the density-dependent growth relationship.

A number of studies suggest the presence of density-dependent growth in longfin populations (Chisnall and Hayes 1991; Chisnall and Hicks 1993; Jellyman et al. 1995; Jellyman 1997). Subsequently, this study estimates a growth term that is fixed across all adult classes in any one year but is able to change *each period* according to the total biomass of the adult stock.

The adopted estimation technique follows the calculation of density-dependent survival for the orange roughy (*Hoplostethus atlanticus*) by Campbell et al. (1993), as this method allows an approximation of the degree to which growth is affected by harvest when more specific biological information is not available. These authors base their approximation on the use of a simulation model to generate equilibrium values of fishable biomass (X_F) and corresponding instantaneous net survival rates ($\xi(X_F)$) for given levels of fishing mortality (F). After identifying levels of X_F and $\xi(X_F)$ for a wide range of fishing mortality values, a relationship between them is estimated using a variety of functional forms until a regression of sufficient accuracy is obtained.

This method rests on a number of intuitive principles. First, harvesting will lower the equilibrium population level. Second, growth rates will be encouraged as harvesting lowers the mean age and thus the average weight of an individual in the population (Campbell et al. 1993; Jennings et al. 2001). These principles lay the foundation for the inverse relationship between the growth rate and the stock level suggested by the operation of density-dependent mechanisms.

The approach of Campbell et al. (1993) is altered to cater for the biology of the longfin and the characteristics of this study. First, the definition of a fishable stock is less appropriate in this present analysis as the management policies to be analysed contain a number of alternative weight restrictions. It therefore follows that the relationship between density-dependent growth and the adult stock (X_{AD}) is estimated. Although weight restrictions will change the way in which this adult stock is affected through harvest, the ages that define this class ($k=[k^a,\dots,k^s]$) do not vary. This is intuitive given the distinct dietary preferences between juveniles and adults (see Section 5.3). Second, the method used to calculate this biomass is different from that adopted by Campbell et al. (1993).

6.6.1 Calculation of the instantaneous growth rate as a function of fishing mortality

The first step in estimating this relationship is deriving an instantaneous growth rate that will change as the exogenous level of fishing mortality is altered. First, in line with equation (6.2), the weight-at-length relationship in symbolic form is:

$$W = \psi \cdot L^{\nu} \tag{6.6}$$

where ψ and ν are both parameters. Likewise in line with equation (6.1), the length-at-age relationship is defined as:

$$L = \alpha + \beta \cdot k \tag{6.7}$$

where α and β are also both parameters and k is age (in years). Substituting equation (6.7) into equation (6.6), a weight-at-age relationship of the following form is obtained:

$$W(k) = \psi \cdot (\alpha + \beta \cdot k)^{\nu} \tag{6.8}$$

where $W(k)$ is the weight of an individual fish of k years old. Schnute (1987) reported that the instantaneous growth rate for a population is related to growth at the mean weight through the following equation:

$$G = \ln\left(\frac{W^{+}}{W}\right) \tag{6.9}$$

where W is the mean weight of an individual fish within the stock and W^{+} is the weight of this fish in the next year. This equation is easily reinterpreted as the instantaneous growth rate (G_F) for the adult eel stock when subject to a given level of fishing mortality (F):

$$G_F = \ln\left(\frac{W_F^{+}}{W_F}\right) \tag{6.10}$$

where W_F is the mean weight of an individual eel within the adult stock for a given level of fishing mortality (F) and W_F^{+} is the weight of an eel of this mean weight (W_F)

in the next period. It is now necessary to derive an expression for W_F. The mean weight of an individual eel in the adult stock for a given level of fishing mortality is defined as:

$$W_F = \frac{\sum_{k=k^a}^{k^s} \left(N_k \cdot W(k)\right)}{\sum_{k=k^a}^{k^s} \left(N_k\right)} \tag{6.11}$$

where N_k is the number of eels in age class k (for $k=[k^a,...,k^s]$) and $W(k)$ is the weight of an individual eel of k years old. It is further postulated that:

$$N_k = R \cdot S_k \tag{6.12}$$

where R is the number of fish that biologically recruit into the stock annually, which is constant because it is assumed that the population is in equilibrium, and S_k is the loss of fish in this recruited platoon occurring through fishing/natural mortality and sexual maturity up to age k. Substituting equation (6.12) into equation (6.11):

$$W_F = \frac{\sum_{k=k^a}^{k^s} \left(R \cdot S_k \cdot W(k)\right)}{\sum_{k=k^a}^{k^s} \left(R \cdot S_k\right)} \tag{6.13}$$

After simplification this yields:

$$W_F = \frac{\sum_{k=k^a}^{k^s} \left(S_k \cdot W(k)\right)}{\sum_{k=k^a}^{k^s} \left(S_k\right)} \tag{6.14}$$

The loss of recruited fish through fishing/natural mortality and sexual maturity up to age k can be further defined as:

$$S_k = e^{\sum_{k=0}^{k} -M} \cdot e^{-F \cdot (k-k^h)} \cdot e^{-\mu(k^m)} \tag{6.15}$$

where $e^{\sum_{k=0}^{k}-M}$ represents the loss of biomass due to natural mortality since biological recruitment, $e^{-F\cdot(k-k^{h})}$ is the loss of biomass due to fishing mortality up to age k for $k=[k^{a},\dots,k^{s-1}]$ (assuming that the fishery is subject to current minimum weight restrictions and thus $k^{a}=k^{h}$), and $e^{-\mu(k^{m})}$ represents the loss of biomass from the migration of males, as a function of the age at which they reach sexual maturity. Substituting equation (6.15) into equation (6.14) provides an expression for the mean weight of an individual fish within the adult stock:

$$W_{F}=\frac{\sum_{k=k^{a}}^{k^{s}}\left(e^{\sum_{k=0}^{k}-M}\cdot e^{-F\cdot(k-k^{h})}\cdot e^{-\mu(k^{m})}\cdot W(k)\right)}{\sum_{k=k^{a}}^{k^{s}}\left(e^{\sum_{k=0}^{k}-M}\cdot e^{-F\cdot(k-k^{h})}\cdot e^{-\mu(k^{m})}\right)} \tag{6.16}$$

It is now necessary to identify the age that corresponds to this mean weight to permit the identification of W_{F}^{+} in equation (6.10). First, equation (6.8) is redefined as an expression for the mean weight of an individual fish for a given level of fishing mortality (W_F) as a function of the average age that corresponds to this intensity of harvest (k_F):

$$W_{F}=\psi\cdot(\alpha+\beta\cdot k_{F})^{v} \tag{6.17}$$

This expression is then solved for k_F. First both sides are divided by ψ and the power (v) is taken to the opposite side to get:

$$(\alpha+\beta\cdot k_{F})=\left(\frac{W_{F}}{\psi}\right)^{\frac{1}{v}} \tag{6.18}$$

Then α is subtracted from both sides, before both sides are divided by β to get:

$$k_{F}=\frac{\left(\frac{W_{F}}{\sigma}\right)^{\frac{1}{v}}-\alpha}{\beta} \tag{6.19}$$

The term (k_F+1) is then substituted into equation (6.17) to find W_F^+:

$$W_F^+ = \psi \cdot (\alpha + \beta \cdot (k_F + 1))^{\nu} \tag{6.20}$$

A simulation model representing each cohort from the first (k^a=11 years) to the last adult age group (k^s=31years) is then developed. This model is built on four of the above equations. First, equation (6.16) calculates the mean weight of an individual fish within the adult stock (W_F) for a given level of fishing mortality (F). This is then used in equation (6.19) to calculate k_F. This is used in equation (6.20) to calculate W_F^+, and equation (6.10) then calculates the corresponding instantaneous rate of growth (G_F) for these values of W_F and W_F^+. While F takes different values to get an understanding of how this changes the mean growth rate, the other values come from previous estimates. In line with earlier discussion, α=58.7, β=35.8, ψ=.00000046, ν=3.28, M=.04, and values for $\mu(k^m)$ are taken from Table 6.1.

6.6.2 Calculation of the adult biomass as a function of fishing mortality

While the above method based on that developed by Campbell et al. (1993) is sufficient for this analysis, the technique used to estimate the fishable stock by these authors is less appropriate. Therefore an alternative method is used. An estimate for the virgin adult stock (X_0) (fish greater than 45cm in length or 220g in weight) is 690 tonnes (Section 6.4). This estimate is used to identify a virgin level of biomass for each individual cohort using the following method. First in equilibrium, the virgin biomass of cohort k (for $k = [k^a,\ldots,k^s]$) is equal to:

$$X_k = R \cdot e^{\sum_{k=0}^{k}(G_k - M - \mu(k^m))} \tag{6.21}$$

where $e^{\sum_{k=0}^{k}(G_k - M - \mu(k^m))}$ is the change in total biomass up to age k through growth (G_k), natural mortality (M), and the loss of migrating males ($\mu(k^m)$). Here G_k is calculated for all discrete growth events using the same method as that adopted for juveniles (Section 6.5). While this rate is clearly density-independent, this formulation is necessary for the estimation of a density-dependent growth relationship for adults. It follows from equation (6.21) that the total virgin adult biomass is:

$$X_{AD} = \sum_{k=k^a}^{k^s} R \cdot e^{\sum_{k=0}^{k}(G_k - M - \mu(k^m))} \tag{6.22}$$

The proportion of the adult stock that consists of cohort k (θ_k) is obtained through dividing equation (6.21) by equation (6.22) and simplifying:

$$\theta_k = \frac{e^{\sum_{k=0}^{k}(G_k - M - \mu(k^m))}}{\sum_{k=k^a}^{k^s} e^{\sum_{k=0}^{k}(G_k - M - \mu(k^m))}} \tag{6.23}$$

This equation is used to form an estimate of θ_k for each adult cohort. The estimate for the weight of the virgin adult stock (X_0=690 tonnes) is then multiplied by each of these proportions to obtain an estimate of the unfished biomass of each cohort ($\theta_k \cdot X_0$). These are then added to the simulation model in the following form:

$$X_{k,F} = \theta_k \cdot X_0 \cdot e^{-F} \tag{6.24}$$

where $X_{k,F}$ is the stock of cohort k for a given level of fishing mortality, $\theta_k \cdot X_0$ is the level of unfished biomass in each adult cohort k, and F is fishing mortality. It follows that the adult stock for a given level of fishing mortality ($X_{AD,F}$) is:

$$X_{AD,F} = \sum_{k=k^a}^{k^s} \theta_k \cdot X_0 \cdot e^{-F} \tag{6.25}$$

This completes the simulation model because as fishing mortality is altered from zero, equation (6.25) changes downward from its virgin estimate (X_0=690 tonnes) and the rate of growth (equation (6.10)) increases. A significant number of pairs of G_F and X_{AD} (with each pair corresponding to an alternative level of fishing mortality) are then obtained and a relationship is fitted to approximate an association between these factors. The form of this association is based on the positive relationship between growth and the difference between the virgin biomass and the adult stock level. The growth rate in period t (G_t), as a function of the adult stock level ($X_{AD,t}$), is therefore:

$$G_t(X_{AD,t}) = \phi \cdot (X_0 - X_{AD,t}) \tag{6.26}$$

where ϕ is a positive parameter. This specification defines that there will be no growth at virgin biomass (which is assumed to be the carrying capacity of the environment), as in the logistic function used to describe population dynamics (Verhulst 1838; Pearl 1930; Schaefer 1957). Yet, as the current adult stock level ($X_{AD,t}$) falls, growth will increase as these larger eels compete less for food and space.

A Cochrane-Orcutt technique (Cochrane and Orcutt 1949) within the SHAZAM econometrics software package (White 1997) is used to estimate this model with first-order autoregressive errors (AR(1)) as serial correlation was detected. The results of this regression (Table 6.3) provide a reasonable description of density-dependent growth (R^2=.946 (3dp)), given that data pertaining to this feature is not available.

Table 6.3. Regression results for the estimation of the growth factor.

Regression type	**Number in sample**	**Dependent variable**	**Independent variable**		**R^2 statistic**
Cochrane- Orcutt method	200	G	**No constant**	**(X_0-X_{AD})**	.946 (3dp)
			(parameter)	.0004	
			(t-statistic)	(247.11)	
			(p-value)	(.000)	

Therefore:

$$G_t(X_{AD,t}) = \phi \cdot (X_0 - X_{AD,t}) \tag{6.27}$$

where ϕ=.0004, X_0=690 tonnes, and $X_{AD,t}$= $\sum_{k=k^a}^{k^s} X_{k,t}$.

This estimated growth factor provides an estimate of how growth may change as stock density fluctuates. This relationship has a stabilising effect, together with stock-related costs and density-dependent recruitment, and therefore aids the solution of the model.

This relationship changes with the protection of given age classes;[11] but the value presented in Table 6.3 (ϕ=.0004) is retained for all policy scenarios to permit direct comparison as significant bias is introduced with the use of alternative values. This bias is most observable in recovery dynamics, as while the higher equilibrium stock level consistent with restrictive weight limits in this simulation model (arising from the short period that animals are vulnerable to the fishery) may have a lower growth *rate* ($\phi(X_0-X_{AD})$) at steady-state, its higher growth *factor* (ϕ) produces infeasible growth, relative to other scenarios, outside of equilibrium. The use of a sole estimate avoids this bias. This estimate is at the lower end of the estimated growth factors for alternative weight restrictions and thus provides a pessimistic view of recovery.

A major limitation of this estimation procedure is that density-dependent growth is independent of age structure in the optimisation model. The use of perfect selectivity in the following model, therefore, presents some difficulty in that exploitation may not be constant across all cohorts and thus a change in the mean age may not stimulate growth as significantly as predicted. This estimation procedure therefore provides an optimistic view of density-dependent growth. An alternative approach would be the use of imperfect selectivity; however, this cannot endogenously determine the harvest/protection of individual year classes since the rate of exploitation is constant across all harvested cohorts. An assumption of perfect selectivity is therefore preferred. This approach follows Conrad (1982) and Moyle (1999) and the recommendations of Anderson (1989).

An optimistic view of sustainable harvest is also provided through assuming that females reach sexual maturity at one fixed age. In reality, rapid growth at low stock densities would decrease the time that eels are available to the fishery and thus decrease the number of year classes from which surplus production may be cropped, although recruitment would increase if harvest were sufficiently controlled during the limited time that longfins were vulnerable to exploitation. Together with the use of perfect selectivity, the sexual maturity of females at one fixed age helps to overcome the

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[11] The estimation of this relationship for alternative weight restrictions is achieved through setting F to zero for protected cohorts in equations (6.16) and (6.25) and then re-estimating the growth factor.

pessimistic view associated with the use of a sole growth factor across all weight restrictions.

6.7 Estimation of a Beverton-Holt stock-recruitment relationship

6.7.1 Background

Stock-recruitment relationships are difficult to estimate for fish species. This difficulty is exacerbated for freshwater eels, as it is impossible to gain detailed information of spawning biomass and corresponding recruitment because of their biology. Nonetheless, the analysis of the recovery and sustainable management of the longfin fishery ideally requires the consideration of stock-recruitment dynamics given the presence of severe recruitment overfishing in this fishery. A non-linear stock-recruitment relationship for the analysed population is therefore derived analytically.

The compensatory Beverton-Holt stock-recruitment relationship (Beverton and Holt 1957) is distinctly different from the assumption of constant recruitment associated with the static yield-effort analysis of Beverton and Holt (1957) and its extensions discussed in Chapter 4. This non-linear specification is very popular within both fish biology (Ricker 1975; Quinn and Deriso 1999; Jennings et al. 2001) and bioeconomics (Kennedy 1992; Campbell et al. 1993), as its simple mathematical form incorporates a powerful biological concept. The Beverton-Holt stock-recruitment relationship is both concave and increasing, but its strength comes from it being one branch of a hyperbola, which asymptotically approaches a maximum recruitment limit (Clark 1976a). This shape portrays the increased mortality of young at high densities.

The Beverton-Holt relationship is selected because the density-dependent mortality of young is very feasible, although little is known of the entire reproductive cycle of this species. Young longfins can spend between fifteen months to two years (see Section 2.3) travelling on ocean currents before reaching New Zealand (Jellyman and Todd 1982) and are capable of limited movement over this time. Competition for food may therefore limit the survival of young. Predation also has a strong effect in both ocean and freshwater environments (see Section 2.3). In particular, birds prey heavily on elvers as they migrate up waterways before settling in home ranges. The suitability of the Beverton-Holt relationship is reinforced when the inappropriateness of alternative specifications is considered. For example, the dome-shaped Ricker curve (Ricker 1975) is based on the cannibalism of young by the spawning stock, which is inappropriate for eels due to their breeding patterns and semelparity.

Ricker (1975) demonstrated how parameters for a Beverton-Holt stock-recruitment relationship could be estimated without spawner and recruitment information. This formulation was for a non-age-structured population in which the entire *fishable* stock was believed to spawn and recruitment was specified as the entry of fish into this class (that is, commercial recruitment). This simplifies the calculation of parameters as the complexity associated with the consideration of age is avoided. The age at which longfins are first vulnerable to gear will vary in this analysis due to the incorporation of different minimum weight restrictions in the management policies to be examined. It is therefore necessary to determine the relationship between spawning biomass and *biological* (rather than commercial) recruitment. Additionally, the total fishable population cannot be used to represent a spawning stock as only a proportion contributes to breeding and these spawners only make a single reproductive contribution. This yields the following specification a spawner-recruitment function, in contrast to those analyses in which the concept of stock and spawner is the same (for example, Ricker 1975). It is important to reiterate here that this function is estimated for spawning females only (see Section 5.3).

6.7.2 Analytical derivation of parameters for the Beverton-Holt spawner-recruitment function

This derivation of the spawner-recruitment function rests on four important assumptions. First, following the nature of the above model, all stock and recruitment units are expressed in tonnes. Second, also as in the above model, the instantaneous rate of growth is assumed to be constant and density-independent for juveniles (0-10 years of age) but density-dependent for adults (11+ years of age). Third, both constant catchability and knife-edge selectivity[12] are assumed. Fourth, it is assumed that the lower Waikato River fishery is managed in its status quo condition; that is, other than

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[12] Knife-edge selectivity denotes the capture of no fish below a given age at first capture (here, $k^a = k^h$) and selection of all fish above this weight at the same degree (Schott 2001).

limited licensing, which is not applicable in this format, only a minimum weight restriction exists.

First, it is necessary to define two distinct forms of recruitment. The first (R_B) describes the weight of animals entering into year class 0 as a function of the spawning stock (X_{SP}). It is assumed that this occurs according to a Beverton-Holt spawner-recruitment relationship:

$$R_B = \left(\frac{X_{SP}}{a \cdot X_{SP} + b} \right) \tag{6.28}$$

where a represents the strength of density-dependent mortality and b is a coefficient determining density-independent mortality.

The second form of recruitment (R_{SP}) is distinctly different from R_B, as it represents the total weight of animals entering into the spawning year class (k^s=31) in equilibrium. In steady-state, for a given level of fishing mortality (F), this is:

$$R_{SP} = \left(\frac{X_{SP}}{a \cdot X_{SP} + b} \right) \cdot e^{\sum_{k=0}^{k^a}(G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{AD}) \cdot (k^s - k^a)} \cdot e^{-(M + \mu(k^m)) \cdot (k^s - k^a)} \cdot e^{-F \cdot (k^s - k^a)} \tag{6.29}$$

where $e^{\sum_{k=0}^{k^a}(G_k - M)}$ is the total change in biomass occurring during the juvenile stage due to growth and natural mortality, $e^{\phi \cdot (X_0 - X_{AD}) \cdot (k^s - k^a)}$ is the total change in biomass occurring during the adult stage due to growth, $e^{-(M + \mu(k^m)) \cdot (k^s - k^a)}$ is the loss of adult biomass due to natural mortality and the migration of males, and $e^{-F \cdot (k^s - k^a)}$ is the loss of adult biomass due to fishing mortality (F).

It is now postulated that Maximum Sustainable Yield (MSY) will occur at the point where recruitment into the spawning year class (equation (6.29)) is parallel to the replacement line (that is, the locus of points defined by R_{SP}=X_{SP}). As the replacement line has a slope of one, MSY will occur where the derivative of equation (6.29) with respect to the spawning stock is equal to unity. Taking the derivative of equation (6.29) obtains:

$$\frac{\partial R_{SP}}{\partial X_{SP}} = \frac{(a \cdot X_{SP} + b - a \cdot X_{SP}) \cdot e^{\sum_{k=0}^{k^a}(G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{AD}) \cdot (k^s - k^a)} \cdot e^{-(M + \mu(k^m)) \cdot (k^s - k^a)} \cdot e^{-F \cdot (k^s - k^a)}}{(a \cdot X_{SP} + b)^2} \tag{6.30}$$

After simplification this becomes:

$$\frac{\partial R_{SP}}{\partial X_{SP}} = \frac{b \cdot e^{\sum_{k=0}^{k^a}(G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{AD}) \cdot (k^s - k^a)} \cdot e^{-(M + \mu(k^m)) \cdot (k^s - k^a)} \cdot e^{-F \cdot (k^s - k^a)}}{(a \cdot X_{SP} + b)^2} \tag{6.31}$$

This expression is now set equal to one and solved for the level of the spawning stock that corresponds to MSY ($X_{SP,MSY}$). Three substitutions are made before this is done. First, $e^{\phi \cdot (X_0 - X_{AD}) \cdot (k^s - k^a)}$ becomes $e^{\phi \cdot (X_0 - X_{MSY}) \cdot (k^s - k^a)}$, where X_{MSY} is the adult stock level at MSY. Second, $e^{-F \cdot (k^s - k^a)}$ becomes $e^{-F_{MSY} \cdot (k^s - k^a)}$, where F_{MSY} is the level of fishing mortality at MSY. Third, X_{SP} becomes the level of female spawning biomass that corresponds to MSY ($X_{SP,MSY}$). Equation (6.31) therefore becomes:

$$1 = \frac{b \cdot e^{\sum_{k=0}^{k^a}(G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{MSY}) \cdot (k^s - k^a)} \cdot e^{-(M + \mu(k^m)) \cdot (k^s - k^a)} \cdot e^{-F_{MSY} \cdot (k^s - k^a)}}{(a \cdot X_{SP,MSY} + b)^2} \tag{6.32}$$

Moving the denominator to the other side and then taking square roots of both sides:

$$a \cdot X_{SP,MSY} + b = \sqrt{b \cdot e^{\sum_{k=0}^{k^a}(G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{MSY}) \cdot (k^s - k^a)} \cdot e^{-(M + \mu(k^m)) \cdot (k^s - k^a)} \cdot e^{-F_{MSY} \cdot (k^s - k^a)}} \tag{6.33}$$

Subtracting b from both sides and dividing both sides by $X_{SP,MSY}$, an expression for a is obtained:

$$a = \frac{\sqrt{b \cdot e^{\sum_{k=0}^{k^a}(G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{MSY}) \cdot (k^s - k^a)} \cdot e^{-(M + \mu(k^m)) \cdot (k^s - k^a)} \cdot e^{-F_{MSY} \cdot (k^s - k^a)}} - b}{X_{SP,MSY}} \tag{6.34}$$

Now it is necessary to identify an expression for b. First, solve equation (6.34) for $X_{SP,MSY}$ and substitute into equation (6.28) to get an expression for recruitment into year class 0 at MSY ($R_{B,MSY}$):

$$R_{B,MSY} = \frac{\dfrac{\sqrt{b \cdot e^{\sum_{k=0}^{k^a}(G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{MSY}) \cdot (k^s - k^a)} \cdot e^{-(M + \mu(k^m)) \cdot (k^s - k^a)} \cdot e^{-F_{MSY} \cdot (k^s - k^a)}} - b}{a}}{a \cdot \left(\dfrac{\sqrt{b \cdot e^{\sum_{k=0}^{k^a}(G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{MSY}) \cdot (k^s - k^a)} \cdot e^{-(M + \mu(k^m)) \cdot (k^s - k^a)} \cdot e^{-F_{MSY} \cdot (k^s - k^a)}} - b}{a} \right) + b} \tag{6.35}$$

After simplification of the denominator, this yields in concise form (using equation (6.34)):

$$R_{B,MSY} = \frac{X_{SP,MSY}}{a \cdot X_{SP,MSY} + b} = \frac{X_{SP,MSY}}{\sqrt{b \cdot e^{\sum_{k=0}^{k^a}(G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{MSY}) \cdot (k^s - k^a)} \cdot e^{-(M + \mu(k^m)) \cdot (k^s - k^a)} \cdot e^{-F_{MSY} \cdot (k^s - k^a)}}} \tag{6.36}$$

It is now postulated that the total adult equilibrium biomass at MSY is:

$$X_{MSY} = \sum_{k=k^a}^{k^s} \left(\left(\frac{X_{SP,MSY}}{a \cdot X_{SP,MSY} + b} \right) \cdot e^{\sum_{k=0}^{k^a}(G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{MSY}) \cdot (k - k^a)} \cdot e^{-(M + \mu(k^m)) \cdot (k - k^a)} \cdot e^{-F_{MSY} \cdot (k - k^a)} \right) \tag{6.37}$$

Taking $\left(\dfrac{X_{SP,MSY}}{a \cdot X_{SP,MSY} + b} \right)$ outside of the summation sign and then substituting equation (6.36) for this expression obtains:

$$X_{MSY} = \frac{X_{SP,MSY} \cdot \sum_{k=k^a}^{k^s} \left(e^{\sum_{k=0}^{k^a}(G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{MSY}) \cdot (k - k^a)} \cdot e^{-(M + \mu(k^m)) \cdot (k - k^a)} \cdot e^{-F_{MSY} \cdot (k - k^a)} \right)}{\sqrt{b \cdot e^{\sum_{k=0}^{k^a}(G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{MSY}) \cdot (k^s - k^a)} \cdot e^{-(M + \mu(k^m)) \cdot (k^s - k^a)} \cdot e^{-F_{MSY} \cdot (k^s - k^a)}}} \tag{6.38}$$

It is now necessary to define that spawning biomass at MSY ($X_{SP,MSY}$) is a certain proportion (θ) of the total eel stock at MSY (X_{MSY}):

$$X_{SP,MSY} = X_{MSY} \cdot \theta \tag{6.39}$$

This proportion is constant since it is assumed that the population is in equilibrium. Solving equation (6.39) for *θ*, substituting it into equation (6.38), and multiplying both sides of the resulting equation by the square root of *b* obtains:

$$\sqrt{b} = \frac{\theta \cdot \sum_{k=k^a}^{k^s} \left(e^{\sum_{k=0}^{k^a} (G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{MSY})(k-k^a)} \cdot e^{-(M+\mu(k^m)) \cdot (k-k^a)} \cdot e^{-F_{MSY} \cdot (k-k^a)} \right)}{\sqrt{e^{\sum_{k=0}^{k^a} (G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{MSY})(k^s-k^a)} \cdot e^{-(M+\mu(k^m)) \cdot (k^s-k^a)} \cdot e^{-F_{MSY} \cdot (k^s-k^a)}}} \tag{6.40}$$

After squaring both sides, an expression for *b* is obtained:

$$b = \frac{\left(\theta \cdot \sum_{k=k^a}^{k^s} \left(e^{\sum_{k=0}^{k^a} (G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{MSY})(k-k^a)} \cdot e^{-(M+\mu(k^m)) \cdot (k-k^a)} \cdot e^{-F_{MSY} \cdot (k-k^a)} \right) \right)^2}{e^{\sum_{k=0}^{k^a} (G_k - M)} \cdot e^{\phi \cdot (X_0 - X_{MSY})(k^s-k^a)} \cdot e^{-(M+\mu(k^m)) \cdot (k^s-k^a)} \cdot e^{-F_{MSY} \cdot (k^s-k^a)}} \tag{6.41}$$

6.7.3 Estimation of parameters for the Beverton-Holt spawner-recruitment function

Estimates of *a* and *b* are identified using the analytical expressions derived for these parameters in the previous section. First, it is necessary to identify all of those parameters appearing in equation (6.41) in order to find *b*. An estimate for the rate of exploitation at MSY (U_{MSY}) is required. The most suitable estimate comes from a stable exploitation rate for New Zealand's freshwater eels identified by Francis and Jellyman (1999), which is 10 percent.

The proportion of the adult stock that consists of spawners at MSY (*θ*) in equilibrium and the adult stock corresponding to this level of exploitation (X_{MSY}) is identified using the simulation model constructed to estimate the rate of density-dependent growth (see Section 6.6). The level of fishing mortality that corresponds to U_{MSY}=.1 is identified using the formula (Ricker 1975):

$$-F_{MSY} = \ln(1 - U_{MSY}) \tag{6.42}$$

This level of fishing mortality is F=-.105 (3dp). Entering this value for F in the simulation model constructed to estimate the rate of density-dependent growth, it is identified that $\theta = .0454$ and X_{MSY}=193 tonnes (rounded to the nearest whole number). This low value for X_{MSY} demonstrates the significant effect that even low levels of exploitation have on longfin populations, resembling the models of Jellyman et al. (2000) and Hoyle and Jellyman (2002). Therefore, using these values, the expression for b in equation (6.41), and data presented previously (see Table 6.1, Section 6.5, and Section 6.6), the value of b is 73.322 (3dp).

In order to calculate the density-dependent mortality parameter, a, an estimate of the spawning stock at MSY ($X_{SP,MSY}$) is identified using equation (6.39) and the estimates of θ and X_{MSY} above. It is identified that $X_{SP,MSY}$=8.76 tonnes. Using this, the value for b provided above, data presented previously (see Table 6.1, Section 6.5, and Section 6.6), and equation (6.34), a is 4.935 (3dp). These values of a and b are parameterised to the standard specification of the Beverton-Holt spawner-recruitment function (Quinn and Deriso 1999) to aid their interpretation. This specification is:

$$R_B = \frac{a^* \cdot X_{SP}}{1 + b^* \cdot X_{SP}} \tag{6.43}$$

where $a^* = \frac{1}{b}$ and $b^* = a \cdot a^*$. The final values of the Beverton-Holt spawner-recruitment function are therefore a^*=.014 (3dp) and b^*=.067 (3dp). Hereafter these parameters are referred to as a and b respectively.

The parameter a is the productivity parameter and is proportional to fecundity. It represents the weight of biological recruitment (in tonnes) per tonne of spawning biomass when this spawning biomass is low (Quinn and Deriso 1999). Its very low value represents the large difference in the weight of individuals at recruitment into the fishery at year 0 (1.46 g) and females at spawning (5.3kg) (see Table A1.1 in Appendix 1).

Additionally, b represents the strength of density-dependence. The low value of b explains the rather gentle slope of the estimated curve (Figure 6.3). Figure 6.3 also illustrates that density-dependence works to force recruitment to asymptotically approach the maximum recruitment limit (*a*/*b*), which is .209 tonnes (3dp).

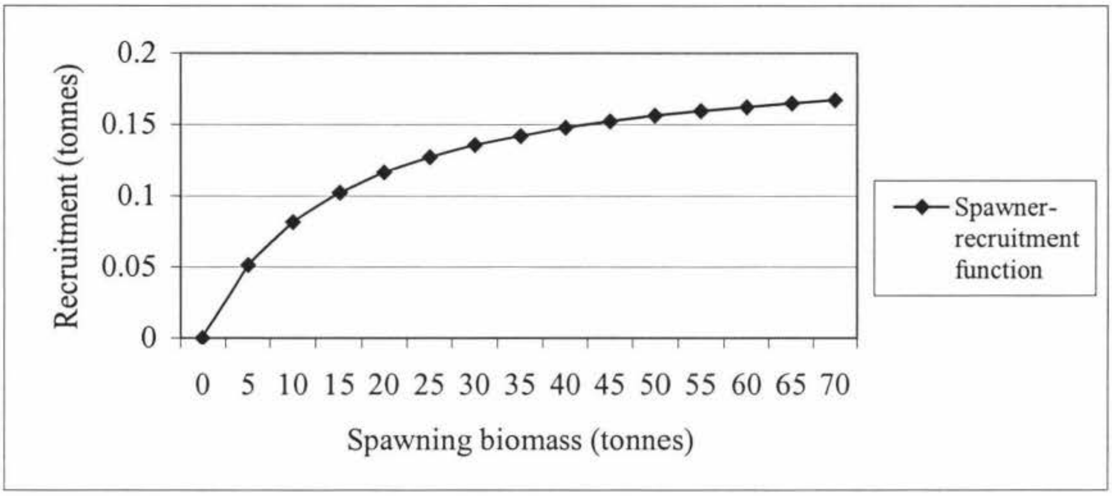

Figure 6.3. The Beverton-Holt spawner-recruitment function.

Although this is a theoretical construct and is based on equilibrium relationships and stock-specific parameter values, its derivation nonetheless allows for some insight into the recovery of longfin populations from recruitment overfishing.

6.8 Specification of the numerical model

This section provides a summary of the parameters and relationships estimated for the studied stock before the numerical model is specified. The cohort references and subscripts used within the numerical model are presented in Table 6.4. The values defining the harvestable year classes (k^h and k^H) will change with alternative weight restrictions.

Table 6.4. Cohort references and subscripts used in the numerical model.

Subscript	Description	Value
ω	Delay between escapement and elver return	ω=2
k^m	The ages at which males reach sexual maturity	k^m=[12, 13,..., 18]
k^h	The youngest cohort vulnerable to gear	k^h=11 in the base model
k^H	The eldest cohort vulnerable to gear	k^H=30 in the base model
k^a	The youngest adult cohort	k^a=11
k^s	The spawning year class	k^s=31

The exogenous parameters used in the base numerical model are provided in Table 6.5.

Table 6.5. Description of exogenous parameters used within this analysis.

Parameter	Description	Value
P_k	Price for harvested cohort k	P_k=\$4350/tonne for k =11-14, P_k=\$4540/tonne for k=15-17, P_k=\$4850/tonne for k=18-20, P_k=\$5950/tonne for k=21-30
c	Cost of harvesting one tonne of eels	c=\$2380.12/tonne
r	Discount rate	r=.05
$\mu(k^m)$	Instantaneous rate of male migration as a function of age	$\mu(12)$=-.015, $\mu(13)$=-.035, $\mu(14)$=-.105, $\mu(15)$=-.114, $\mu(16)$=-.149, $\mu(17)$=-.057, $\mu(18)$=-.025
X_0	Virgin biomass	X_0=690 tonnes
$\sum_{k=0}^{k^a}(G_k - M)$	Survival rate for juveniles	$\sum_{k=0}^{k^a}(G_k - M)$ =4.65
M	Instantaneous annual rate of natural mortality	M=.04
ϕ	Density-dependent growth factor	ϕ=.0004
a	Parameter for spawner-recruitment function	a=.014
b	Parameter for spawner-recruitment function	b=.067

The numerical model consists of the substitution of those relationships estimated in this chapter into the analytical model presented in Section 5.4, together with some modification to permit the description of alternative management strategies. First, it is necessary to restrict the harvest ($Y_{k,t}$) of all *adult* cohorts *below* the minimum age at first capture (k^h) or *above* the maximum age at capture (k^H) to zero with alternative weight restrictions. Second, to explore the optimality of alternative quota levels, it is necessary to specify the "TAC" constraint:

$$X_{AD} \cdot U_q \geq Y_t \tag{6.44}$$

where U_q is the maximum exploitation rate permitted at time t (for example a TAC of 20 percent exploitation is defined as U_q=.2), X_{AD} is the total adult stock ($X_{AD} = \sum_{k=k^a}^{k^s} X_{k,t}$), and Y_t is total harvest ($Y_t = \sum_{k=k^h}^{k^H} Y_{k,t}$). Third, the standard intuitive feasibility constraints also apply; that is, the total adult stock may never surpass its virgin level, stock and harvest levels are always non-negative, and the harvest of cohort k at time t must never be greater than the stock of cohort k at time t.

The numerical model takes the form:

Maximise:

$$\pi = \sum_{t=0}^{\infty} \rho^t \cdot \left(\sum_{k=k^h}^{k^H} Y_{k,t} \cdot \left(P_k - c \cdot \left(\frac{Y_{k,t}}{X_{k,t}} \right) \right) \right) \tag{6.45}$$

Subject to:

$$X_{k^a,t+k^a+\omega} = \frac{a \cdot X_{k^s,t}}{1 + b \cdot X_{k^s,t}} \cdot e^{\sum_{k=0}^{k^a} (G_k - M)} \tag{6.46}$$

$$X_{k+1,t+k^a+\omega+1} = X_{k,t+k^a+\omega} \cdot e^{\phi \cdot (X_0 - X_{AD,t+k^a+\omega}) - M - \mu(k^m)} - Y_{k,t+k^a+\omega} \tag{6.47}$$

for $k=[k^a,\ldots,k^{s-1}]$.

$$Y_{k,t} = 0 \tag{6.48}$$

for all adult cohorts below k^h or above k^H.

$$X_{AD} \cdot U_q \geq Y_t \tag{6.49}$$

Together with the feasibility constraints:

$$X_0 \geq X_{AD} \tag{6.50}$$

$$X_{k,t} \geq 0 \tag{6.51}$$

$$Y_{k,t} \geq 0 \tag{6.52}$$

$$X_{k,t} \geq Y_{k,t} \geq 0 \tag{6.53}$$

Together with the definitions of the adult stock and total harvest to fully specify the numerical model:

$$X_{AD} = \sum_{k=k^a}^{k^s} X_{k,t} \tag{6.54}$$

$$Y_t = \sum_{k=k^h}^{k^H} Y_{k,t} \tag{6.55}$$

There is a minimum of thirteen initial conditions required as the delay encompassed in the first two constraints (equations (6.46) and (6.47)) is thirteen years ($k^a+\omega$) (Table 6.4). The initial condition specified for thirteen years is the spawning biomass consistent with an estimate of historical exploitation (20 percent) (Jellyman et al. 2000), as this establishes that recruitment in the first thirteen periods is beyond the influence of current management because of the delay between spawning and the vulnerability of corresponding recruits to the fishery. Time in the model begins at t=-13 to accommodate these initial conditions and continues in annual intervals. Only the results from t=0 onwards are reported, as output for the first thirteen years (t=[-13,…,-1]) is disregarded due to its sole use for establishing initial recruitment.

Additionally, equilibrium cohort stock levels consistent with a 20 percent exploitation rate are included at t=0 as initial conditions. These are estimated using the simulation model constructed to identify the density-dependent growth factor (Section 6.6). The inclusion of these cohort stocks decreases solution time, as a number of periods are not required for the model to calculate year class levels before meaningful output can be produced.

The 20 percent rate of harvest used for these initial conditions effectively described historical exploitation in one waterway (Jellyman et al. 2000), although these authors posited that this is likely to be the maximum level encountered. Using this initial condition allows for greater realism when comparing the suitability of different management policies, as the number of unexploited fisheries is declining and therefore only in exceptional cases will managers be faced with forming regulatory policies for virgin longfin fisheries. The implications of the dynamics of recovery identified by this model will also be widely applicable, given that all stocks should be in a more healthy state than this fishery because of the pessimistic view provided by the magnitude of this rate.

In summary, these initial conditions specify that the initial adult population will be that consistent with a 20 percent exploitation rate and that recruitment for the first thirteen years will be fixed at that level corresponding to spawning biomass at this rate of harvest, as this latter effect is beyond the influence of current management.

6.9 Solution procedure

This model is formulated in an NLP framework and is solved using the CONOPT2 solver in the General Algebraic Modelling System (GAMS) (Brooke et al. 1988; Jefferson and Boisvert 1989). The specific version used is GAMS 2.25.092. The GAMS module is particularly suited to solving this model because its programming language and attached solvers allow for the concise description and solution of even large and complex problems.

The CONOPT2 solver adopts a feasible path approach based on a Generalised Reduced Gradient (GRG) algorithm (Drud 1985, 1992). For detailed information on this procedure, see GAMS (2002a). Although many optimisation problems in fisheries utilise the MINOS solver (Larkin and Sylvia 1999; Schott 2000; Lee et al. 2000), CONOPT2 is suited to solving models of the kind used in this analysis due to the presence of non-linear constraints (GAMS 2002b).

The CONOPT2 solver had difficulty in identifying an optimum on a number of occasions, although the calculated solution was feasible in all cases. Re-optimisation of these models by the GAMS Development Corporation (GAMS 2002c) identified that the solutions provided by CONOPT2 were indeed optimal, but no solvers, except the most recent versions of CONOPT3 and MINOS, were able to prove them so. The results of this re-optimisation motivated the GAMS Development Corporation to state, "... you can be fairly certain that although CONOPT2 cannot prove that the solution is optimal, it in fact is" (GAMS Development Corporation, *pers. comm.*, 7 August 2002). All feasible maxima that are non-optimal are therefore treated as optimal after it is ensured that no alternative optima exist. The presence of these alternative solutions on the decision hypersurface is determined through the use of different initial conditions.

To solve this infinite horizon problem using GAMS software, it is truncated to a finite horizon of significant length (seventy-eight years), a method also adopted by Rowse (1995) and Schott (1999, 2000). The length of this period permits an analysis incorporating sixty-five years of meaningful output, given that an initial thirteen-year period is required to allow for the model to successfully establish (see Section 6.8). Although a longer period may have been preferable, the high number of cohorts involved significantly increased solution time without the addition of any greater insight.

Truncation to a finite horizon introduces bias, as unsustainable harvest in the last periods is perfectly rational given the "end of the world" scenario depicted. Divergences from the steady-state optimum during this time are disregarded as artefacts from truncation. This approach is more suitable than specifying a terminal condition (for example a minimum level of spawning biomass) (Yamauchi et al. 1997) as a lack of information prevents the identification of a suitable candidate and no terminal period will exist in reality.

The GAMS programme written for the base run of the model is presented in concise form in Appendix 4 and with a detailed description of the programming language and approach taken to solution (for example model scaling) in Appendix 5.

6.10 The analysis of alternative management strategies

The alternative management policies to be optimised in the numerical model are selected using a variety of criteria. First, various policies are adopted to provide insight into the present sustainability of the fishery and the optimal harvest pattern for the status quo regulatory programme. Second, a combination of TAC levels and weight restrictions are simulated in order to identify their comparative optimality.

6.10.1 Policy scenarios reflecting the current state of the fishery

The first policy involves the optimisation of the base model; that is, this framework in its original form and with its original parameter values. This provides insight into the optimal pattern of exploitation for the sole owner when the fishery is subject to existing regulations and provides a baseline to which alternative policies are compared.

The second policy portrays open-access conditions in this model through the use of a discount rate that is tending to infinity (Clark 1976a, 1985). This approach replicates open-access through ascribing no weight to future profits accruing from resource use. This policy scenario is necessary because significant competition for fish exists within the current regulatory scheme imposed in the North Island fishery.

The third policy investigates the future ramifications of current quota-setting methodology through fixing exploitation to current levels. The adopted estimate of the current exploitation rate is 20 percent (Jellyman et al. 2000). This rate represents a ceiling to commercial fishing and thus the outcomes of this scenario should be regarded as pessimistic. This approach may be preferable to one that portrays more optimism as it provides more conservative insight into the management of this fishery in agreement with the importance of a precautionary approach to the formulation of appropriate regulatory strategies for this species (Hoyle and Jellyman 2002).

6.10.2 Alternative management policies

The following policies reflect combinations of TACs and weight restrictions that would make possible alternatives to the existing management strategy. These policies can be grouped into three categories, corresponding to unconstrained harvest and base TACs of 5 and 10 percent. The first (5 percent) identifies the relative efficiency of the lower bound of optimal exploitation rates identified by Hoyle and Jellyman (2002). The second (10 percent) investigates the effects of a TAC set at the stable rate identified by Francis and Jellyman (1999). This is also close to the higher bound of estimates of an optimal exploitation rate (8 percent) identified by Hoyle and Jellyman (2002). The unconstrained harvest scenario identifies the optimal TAC level when no limit is placed on exploitation.

A number of minimum and maximum weight restrictions are used within each harvest scenario; that is, for the two TAC options and for unconstrained harvest. Minimum weights of .5kg, 1kg, and 1.5kg are used. Their use follows the need to investigate the effects of alternative weight limits given the current pricing schedule (Section 6.3.1), recommendations for substantial increases in minimum size restrictions (Chisnall et al. 2002), and the use of similar weights by Hoyle and Jellyman (2002). Maximum weights of 2kg, 3kg, and 4kg are also used. This follows the use of similar weights by Hoyle and Jellyman (2002), permits the portrayal of a range that represents various levels of conservativeness, and investigates the effectiveness of the 4kg limit imposed in the South Island. Additionally, a scenario incorporating no maximum weight restriction is included to identify how this influences the efficient management of this fishery.

The age groups available to the fishery for each set of simulated weight restrictions are presented in Table 6.6. These are identified using the mean weight for each cohort presented in Table A1.1 in Appendix 1.

Table 6.6. The year classes available to the fishery for each set of analysed weight restrictions.

Minimum weight (kg)	**Maximum weight (kg)**			
	2	**3**	**4**	**No max.**
.5	15 to 22 yrs	15 to 25 yrs	15 to 28 yrs	15 to 30 yrs
1	18 to 22 yrs	18 to 25 yrs	18 to 28 yrs	18 to 30 yrs
1.5	21 to 22 yrs	21 to 25 yrs	21 to 28 yrs	21 to 30 yrs

These weight restrictions are applied in every possible combination across all harvest scenarios to provide a detailed picture of the relative optimality of potential management strategies. The combination of all harvest and weight restrictions represents a total of thirty-six scenarios. These strategies are entered into the model through manipulation of the summation term for *k* in the objective function (equation (6.45)) and adjustment of equations (6.48) and (6.49).

7. RESULTS AND DISCUSSION

7.1 Introduction

In this chapter the results of the model described in Chapter 6 are presented, interpreted, and discussed in terms of their implications for the management of the longfin eel fishery. It is demonstrated that managing this resource to maximise economic efficiency diverges from those policies that sustain significant levels of spawning biomass (Jellyman et al. 2000; Hoyle and Jellyman 2002).

First in this chapter, the implications of the base model are explored. This is followed by a discussion of the results for the open-access and status quo scenarios. Model output for a number of alternative management scenarios is then presented. The sensitivity of the model is tested before the implications of model output for an appropriate management strategy are discussed.

7.2 Models providing insight into existing longfin eel fishery conditions

7.2.1 Implications of the base model

This model is formulated to mimic the current management of the North Island longfin fishery in that only a minimum weight restriction (220g) (consistent with $k^a = k^h$) is represented. This formulation introduces the important components of an efficient management strategy, which are discussed in further detail in Section 7.3.

The optimal recovery policy requires that harvests be maintained at low levels for the first five years (Figure 7.1). This permits spawning biomass to increase to its steady-state value (three tonnes) rather rapidly (six years). This indicates that the implementation of moratoriums or restrictive harvest controls for a short period would be necessary if the fishery were to rapidly recover from its exploited position. The equilibrium exploitation rate is moderate (14.1 percent), but significantly lower than an estimate of 20 percent used to describe historical harvest levels by Jellyman et al. (2000).

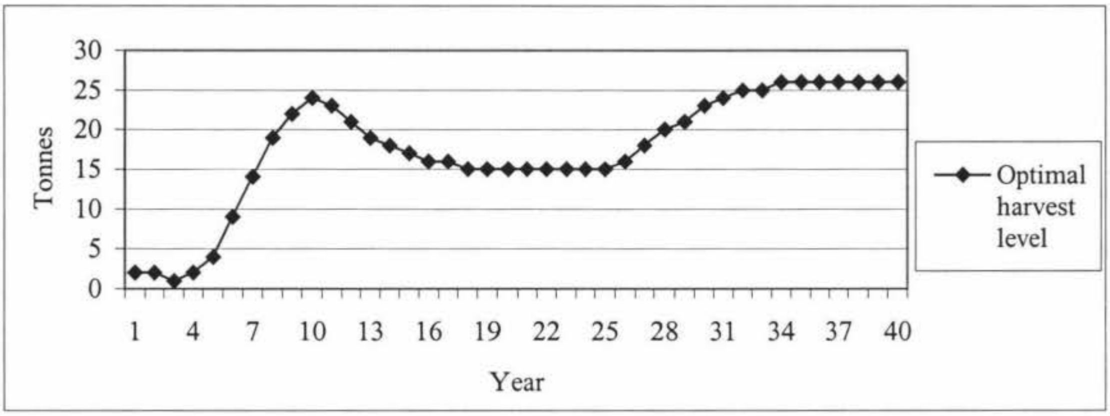

Figure 7.1. Optimal harvest levels for the base run of the model.

Harvest is restricted to the most valuable age classes (year classes 21 to 30, which are all female) (see Table 6.2 for relative monetary values), with exploitation of these cohorts increasing steadily until the equilibrium age structure (Figure 7.2) is obtained.

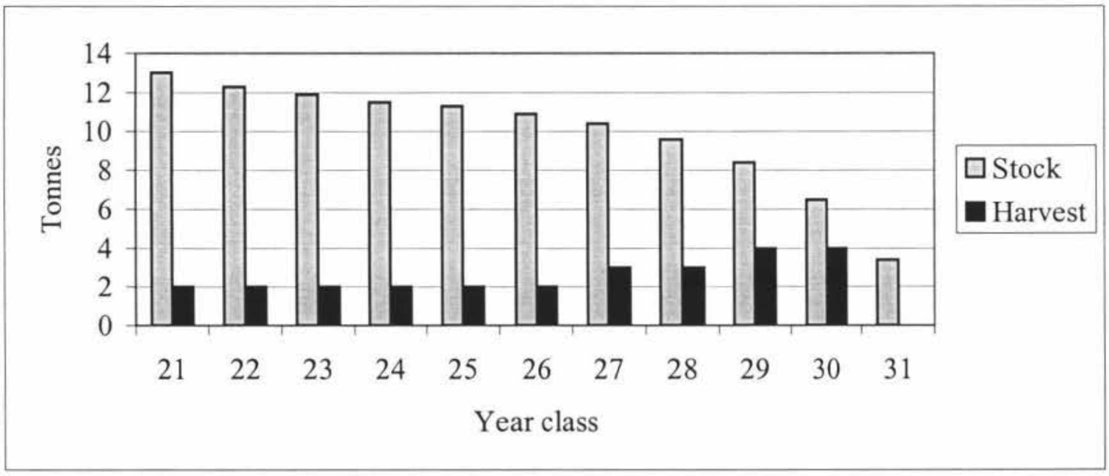

Figure 7.2. Equilibrium stock and harvest for each of the most valuable year classes and the spawning cohort for the base model.

The intuitive concentration of harvest on the most valuable cohorts within a population has been demonstrated previously (Gatto et al. 1982; Moyle 1999). Yet this finding has a number of important implications for the longfin fishery. First, the base model output demonstrates that lower maximum weight limits recommended by various authors (Chisnall and Hicks 1993; Chisnall et al. 2002; Hoyle and Jellyman 2002) may limit the harvest of the most valuable year classes, therefore decreasing the potential value of the resource, given that it will involve a significant divergence from the optimal age structure. Second, no harvest of less valuable year classes (220g to 1.4kg, or ages 11 to 20 within this model) indicates that growth overfishing may be overcome through increasing the minimum weight restriction to 1.5kg. This demonstrates the inefficiency of current exploitation, as the majority of harvested eels are now less than 500g in weight (Beentjes 1999). Although this high minimum weight restriction would protect males from exploitation because all spawn beneath this weight (Todd 1980), the harvest of females must be limited to moderate levels if spawning biomass is to be maintained as this weight limit merely delays the harvest of this longer-living gender.

The long-run equilibrium stock level is much lower than the virgin biomass (X_0=690 tonnes) (Figure 7.3). This is consistent with low levels of equilibrium recruitment arising from weak spawning classes. These spawning cohorts are maintained at low levels since investment in these will impact on the exploitation of the most valuable cohorts (Figure 7.2), given semelparity. Even though spawning year classes are weak, rapid recovery is promoted through increased growth due to the lessening of competition among adults.

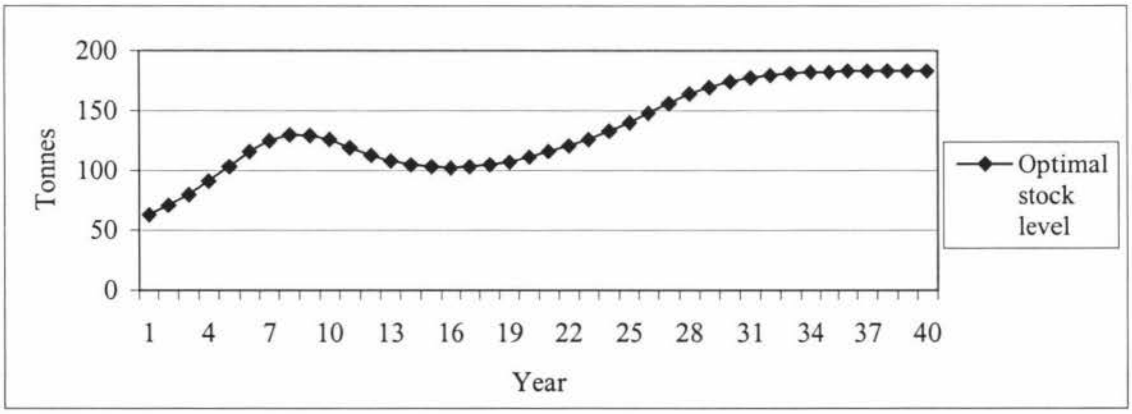

Figure 7.3. Optimal stock levels for the base run of the model.

The base run of the model demonstrates that the greatest economic gain may be made from this difficult situation, whereby more valuable cohorts are those approaching spawning, by decreasing the time that females are available for harvest through the use of a higher minimum weight restriction and then decreasing the degree of exploitation within the fishable population.

7.2.2 Implications of the open-access model

A discount rate tending to infinity is used to simulate open-access conditions (Clark 1976a, 1985). From year two, the population is heavily harvested over two years until only minor yields may be cropped annually (Figure 7.4). The population reaches equilibrium from year fifteen onwards, where the optimal stock remains around four tonnes and harvest around one tonne. This harvest profile is consistent with the predictions of Gordon (1954), as Net Present Value (NPV) is zero under this scenario.

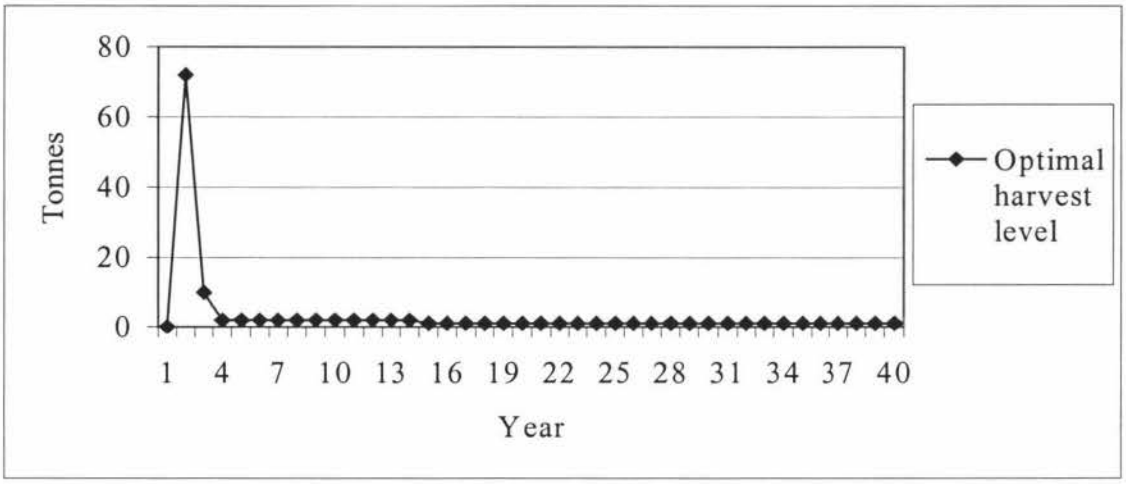

Figure 7.4. Optimal harvest levels for the model depicting open-access conditions.

This model demonstrates that no eel fishers have an incentive for the sustainable harvest of the eel resource in the absence of rights-based regulation. This indicates the importance of formulating an effective rights-based management system in this fishery, because although a high number of surveyed harvesters expressed an interest in its sustainable management (see Question 16E in Appendix 3), presently there is no incentive for them to exhibit such behaviour, except on some private properties where they have been granted sole access. The problems associated with the implementation of an ITQ management system for the longfin fishery have been discussed previously (Section 3.6.5).

7.2.3 Implications of maintaining current exploitation

This model investigates the maintenance of a 20 percent exploitation rate and therefore the effect of current TACs, given quota-setting methodology. It is assumed that the sole owner will maximise profit by harvesting his/her full quota. With the fixing of harvest at thirteen tonnes (20 percent of the initial population) compared to an equilibrium harvest for the base case of twenty-six tonnes, the population experiences very little movement around the initial stock size (seventy-one tonnes). The inefficiency of a high rate of harvest without permitting recovery is clear, as this policy reports the only negative Net Present Value (NPV) of all scenarios, reporting a loss of $13,900, compared to the base run of the model that returns $881,500. Equilibrium spawning biomass decreases under this fishing regime to .3 tonnes. These factors demonstrate that the maintenance of heavy harvest will sustain yield below higher, more profitable levels, increase costs as eels become more expensive to fish as stocks fall, and place the population at increased risk of collapse, given the effect of exploitation on spawning biomass.

This model indicates a number of important factors. First, the use of past harvest to calculate TACs has severe negative implications for both spawning biomass and efficiency. Second, the conclusions of the model are sensitive to the inclusion of deterministic recruitment, density-dependent growth, and stock-dependent costs, as these promote stability at low stock levels and thus help to prevent extinction of the analysed population. This indicates the importance of investigating the soundness of these assumptions because while the first is a significant abstraction, the final two have yet to be sufficiently validated and measured. Third, this model demonstrates the importance of formulating stock assessment procedures for longfin populations. While this is problematic, this information would allow quotas to adjust with stock size in order to keep rates of exploitation at sustainable levels and therefore decrease the sensitivity of these populations to collapse.

7.3 Optimal management of the longfin fishery

The output of the base model suggests three important factors for the efficient management of the longfin fishery: a high minimum weight restriction, no maximum weight limit, and decreased harvest. This section presents the key findings of the analysis of alternative management strategies in terms of these factors. These results demonstrate the magnitude of inefficiency that accrues to constraining the implementation of the optimal policy. Key findings are reported here, while detailed information on output for all analysed policies is presented in Appendices 6 to 9.

7.3.1 Importance of a higher minimum weight restriction

It is a feature of model output that levels of discounted profit for different minimum weight restrictions (for a given level of exploitation and maximum weight limit) are very similar (Figure 7.5).[13] This arises from the concentration of harvest on those cohorts that belong to the most valuable year classes, regardless of what minimum weight limit is in place. This indicates the theoretical importance of a minimum weight restriction set at the lower bound of the most valuable year classes (1.5kg).

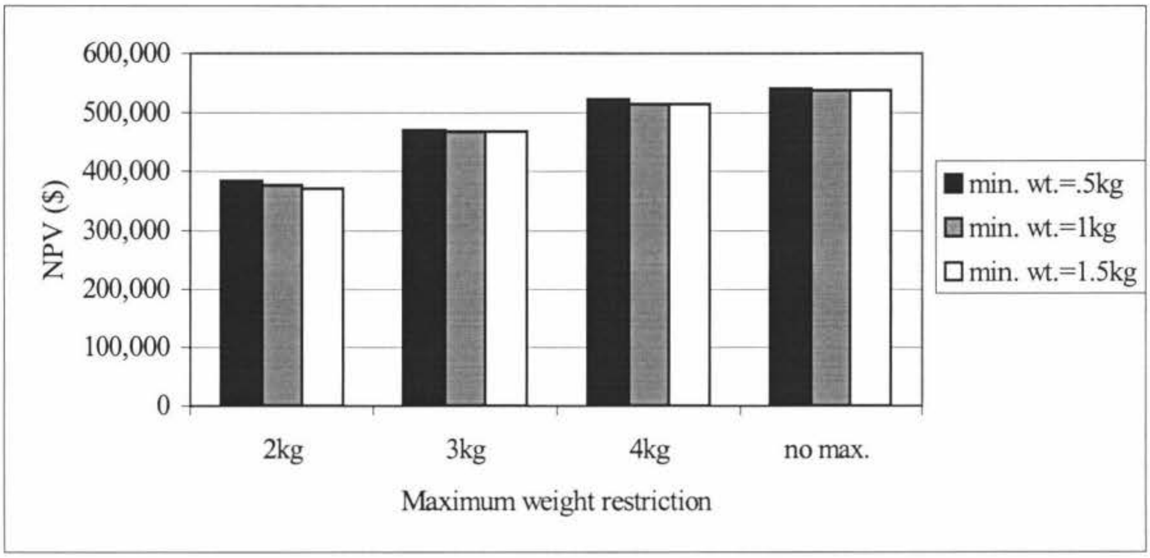

Figure 7.5. Levels of discounted profit for varying minimum weight restrictions across all analysed maximum weight limits for a TAC of 5 percent.

Less valuable cohorts are only exploited on two occasions. First, the minor superiority of those policies with lower minimum weight restrictions (Figure 7.5) follows the harvesting of year classes beneath the lower bound of the most valuable cohorts (1.5kg) as the terminal period approaches. This behaviour is rational since the harvesting of all available year classes at this time will have no future cost. The minor superiority of lower minimum weight restrictions is ignored because no obvious terminal period exists in reality, and thus exploitation of only the most valuable cohorts would continue

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[13] This effect is observable over all levels of exploitation and can be identified in greater detail in the Net Present Values (NPVs) for all alternative management strategies in Table A6.1 in Appendix 6.

indefinitely within the optimal strategy. Second, less-valuable year classes are also harvested when a maximum weight limit of 2kg is in place because of the significant restriction on the number of cohorts vulnerable to the fishery. For example, this maximum weight restriction prevents the harvest of 80 percent of the most valuable year classes. This effect is of low importance because of the low levels of NPV accruing to these policies relative to those adopting higher maximum weight limits (Figure 7.5).

The allocation of property rights should theoretically prevent the harvest of fish of sub-optimal size as fishers maximise the value of quota. Nonetheless, a minimum weight restriction would be necessary within the optimal policy to prevent growth overfishing as fishers would have an increased incentive to exploit younger year classes if quotas were set too high, for example because of poor stock data, and larger eels were scarce.

A minimum weight limit of 1.5kg would prevent the capture of the majority of shortfins, as around 95 percent (Jellyman 1993) spawn beneath this weight and this species, like the longfin, is semelparous. Gear also cannot select between species. Severely limiting the exploitation of shortfins would have serious implications for profit because over half of total harvest consists of this species (60 to 65 percent) (Annala et al. 2001). This result indicates that while 1.5kg is the theoretical optimum in longfin fisheries, this minimum weight restriction is unsuitable for wide application. This finding does demonstrate, though, that a higher minimum weight limit will increase efficiency, with potential gains increasing in the level of minimum weight imposed.

7.3.2 Importance of user cost in determining the optimal harvest profile

The optimality of concentrating harvest on older eels is consistent with the opportunity cost of harvesting them at a sub-optimal size. This is identified in Sections 5.6 and 5.7, where the partial solution of the analytical model indicates that user cost and user benefit play a significant role in the determination of an optimal management strategy. This effect is pronounced as longfins are long-lived and thus exploitation of a young year class in the current period will affect the fishery for many years (especially when a higher or no maximum weight restriction is in place) as this cohort moves sequentially through the population before it eventually spawns. The importance of user cost and user benefit in determining the age composition of harvest is demonstrated in that harvesting biomass in a younger, less valuable year class will:

1. remove the option of exploiting this biomass in the future, once it has grown
2. decrease the potential value of a given unit of biomass as it is denied the chance to move into more valuable year classes
3. decrease the contribution of this biomass to lowering costs in the future through increasing stock levels
4. decrease the contribution of this biomass to spawning, and
5. increase the growth of surviving biomass as stock density decreases.

The first four effects (1 to 4) collectively outweigh the growth effect (5) across all management strategies, as harvest builds gradually as a female year class approaches spawning (Figure 7.6). The cropping of the terminal *harvestable* cohort (for example, year 30 in Figure 7.6) is promoted above that for other year classes, given that the first three components of user cost listed above (1 to 3) no longer accrue to it, because in the next year it is no longer available to the fishery. The partial cropping of the most valuable year classes, with exploitation increasing as spawning draws nearer, is observable within all simulated policies, demonstrating the importance of user cost to an optimal management strategy for a long-lived, semelparous population.

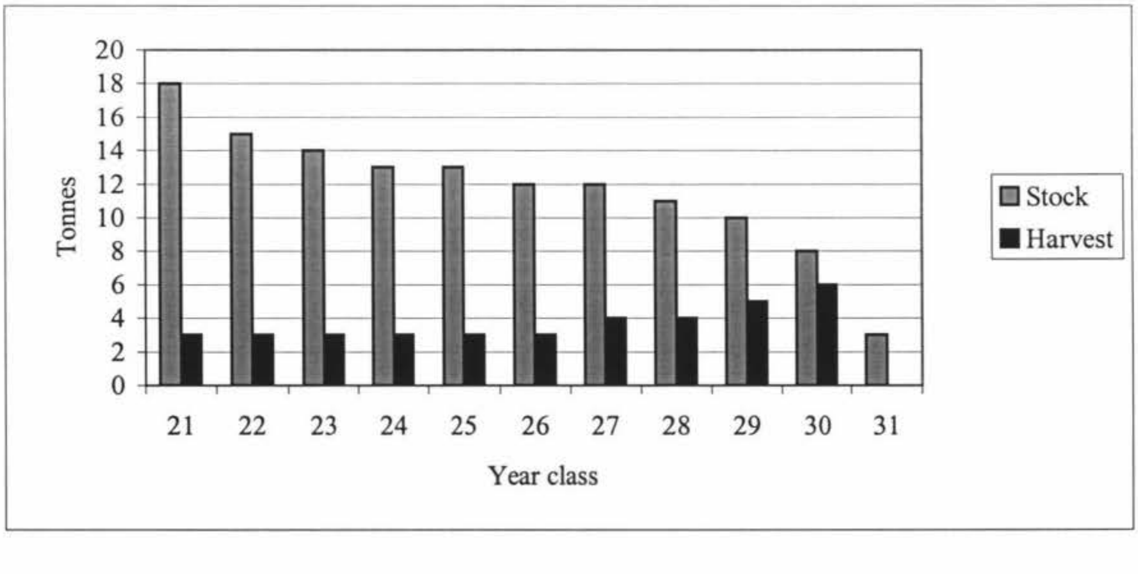

Figure 7.6. Equilibrium stock and harvest levels for each of the most valuable year classes and the spawning cohort for a minimum weight limit of 1.5kg, no maximum weight restriction, and no TAC.

Lowering the price received for harvested eels affects the optimal management strategy through decreasing the future value of biomass. This is observable in one of the rules for optimal management derived in the partial analytical solution of the model in Section 5.6, as a decrease in price lowers the discounted value of an additional unit of biomass in the next cohort in the next period ($\rho \cdot \lambda_{k+1,t+1}$) in the equation:

$$P_k - \frac{\partial C(X_{k,t}, Y_{k,t})}{\partial Y_{k,t}} = \rho \cdot \lambda_{k+1,t+1} \tag{7.1}$$

for $k=[k^h,\ldots k^H]$.

The strength of the price effect on the optimal management strategy is tested to identify its importance relative to the other components of user cost listed above. The mean price (P_k=\$4922.5) from the current harvesting schedule (Table 6.2) is used for all year classes to investigate this factor. While a constant price makes some minimal fishing of younger cohorts attractive (Figure 7.7) and intuitively decreases NPV (which falls by 17.7 percent under this scenario), the general nature of the optimal age structure of equilibrium harvest (partial cropping with exploitation increasing as spawning draws nearer) is particularly robust to this modification, as the year classes approaching spawning continue to constitute a significantly higher proportion of total harvest. In fact, 81 percent of harvest consists of the previously most valuable cohorts (year classes 21 to 30) when prices are constant across all age groups.

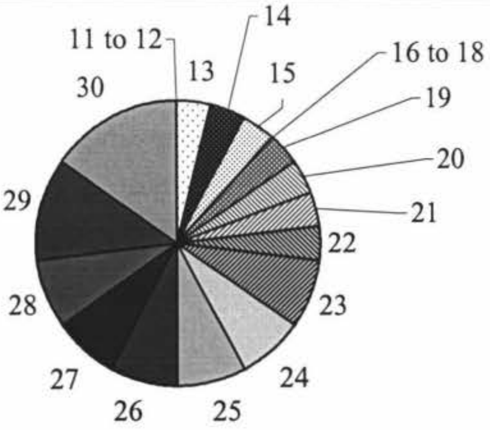

Figure 7.7. The optimal age structure of equilibrium harvest when the value of a harvested tonne of eels from each year class is the same (\$4922.50).

This identifies that although price is important, total harvest should consist of larger longfin females even when no premium is paid for these eels. This reflects the declining levels of user cost accruing to the exploitation of eels approaching spawning, as the benefits of maintaining year class strength for both future harvest and lower costs become less important as these animals approach the time at which they leave the fishery. This finding reinforces the importance of a higher minimum weight restriction to the optimal management of this fishery, as this indicates the particular robustness of this conclusion to the removal of the price effect.

It may be proposed that focusing exploitation on the most valuable cohorts would lead to a change in demand elasticities so that smaller eels become more valuable relative to larger animals; however, this will not occur because New Zealand's eel exports comprise only a small proportion of world supply (see Section 6.3.1). The robustness of model output to a constant price also indicates that the influence of such an effect, if it did exist, is likely to be small.

These results demonstrate that the majority of total harvest should ideally consist of animals approaching spawning because of their higher value in terms of decreased user cost. This demonstrates the importance of user cost to the determination of the optimal age structure of harvest for a semelparous, long-lived species.

7.3.3 Importance of no maximum weight restriction

Maximum weight limits severely constrain the maximisation of economic efficiency within the longfin fishery. Figure 7.8 illustrates that NPV increases as maximum weight limits bind less strongly and permitted harvest increases.[14] The relative optimality of having no maximum weight limit (Figure 7.8) stems from harvest levels that are higher than those accruing to the use of other maximum weight restrictions. For example,

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[14] These effects are observable over all minimum weight restrictions and can be identified in greater detail in Table A6.1 in Appendix 6.

equilibrium harvest is between 12 and 28 percent higher for policies with no maximum weight restriction compared to a maximum weight of 2kg, *ceteris paribus.*[15]

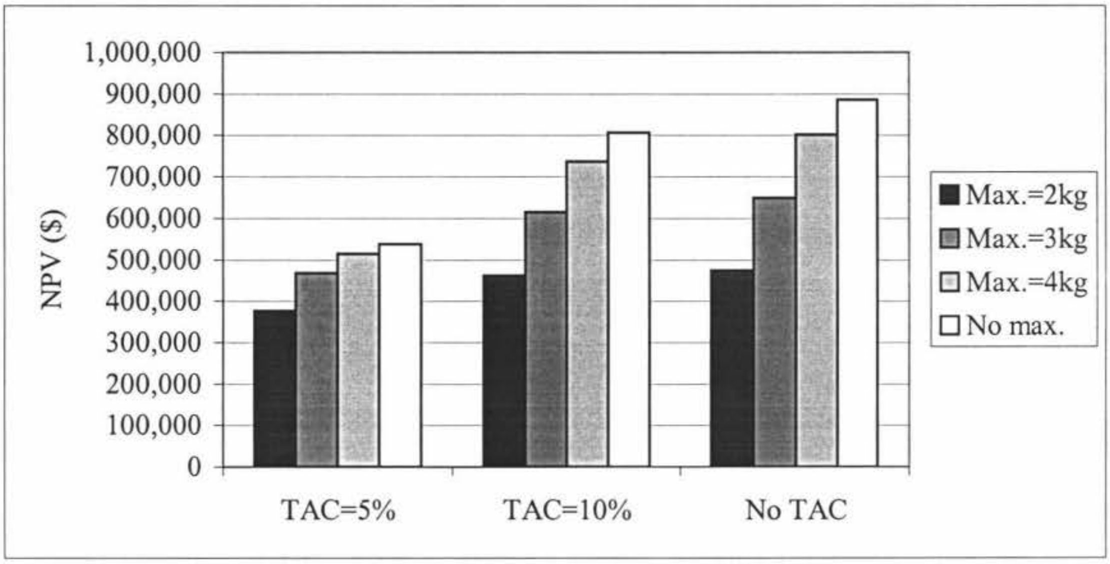

Figure 7.8. Levels of discounted profit for varying maximum weight restrictions and levels of permitted harvest for a minimum weight limit of 1kg.

Harvest entirely removes surplus production at any sustainable equilibrium (Clark 1976a; Conrad 1999), and since the annual growth of the greatest number of the most valuable cohorts is available for cropping when no maximum weight restriction is in place, it is intuitive that greater levels of harvest may be taken at steady-state. The fact that harvest is higher, but not greatly so, reflects that even though a higher number of cohorts are available, younger age groups are not heavily exploited given the user cost accruing to their removal.

Jellyman et al. (2000) and Hoyle and Jellyman (2002) reported that maximum weight limits of 2kg or lower were required to significantly increase spawning biomass within the longfin fishery, yet these lower restrictions have severe negative implications for economic efficiency. Model output demonstrates that NPV nearly doubles with the use of no maximum weight restriction, relative to a limit of 2kg, when harvest is unconstrained (Figure 7.8). The relative optimality of no maximum weight limit also

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[15] See Table A7.1 in Appendix 7 for harvest levels for all alternative management strategies.

indicates the inefficiency of an upper weight limit of 4kg used in the South Island, as this prevents the harvest of a number of the most valuable year classes. This finding complements that of Hoyle and Jellyman (2002), who reported that this restriction is also ineffective as a tool for protecting spawning biomass.

Semelparity strongly influences the determination of an optimal management strategy. Since all harvested eels have yet to contribute to spawning, an increased level of exploitation directly decreases future spawning biomass and therefore recruitment. It follows that while harvest, and consequently NPV, increases as maximum weight restrictions bind less strongly; spawning biomass decreases (Figure 7.9).[16]

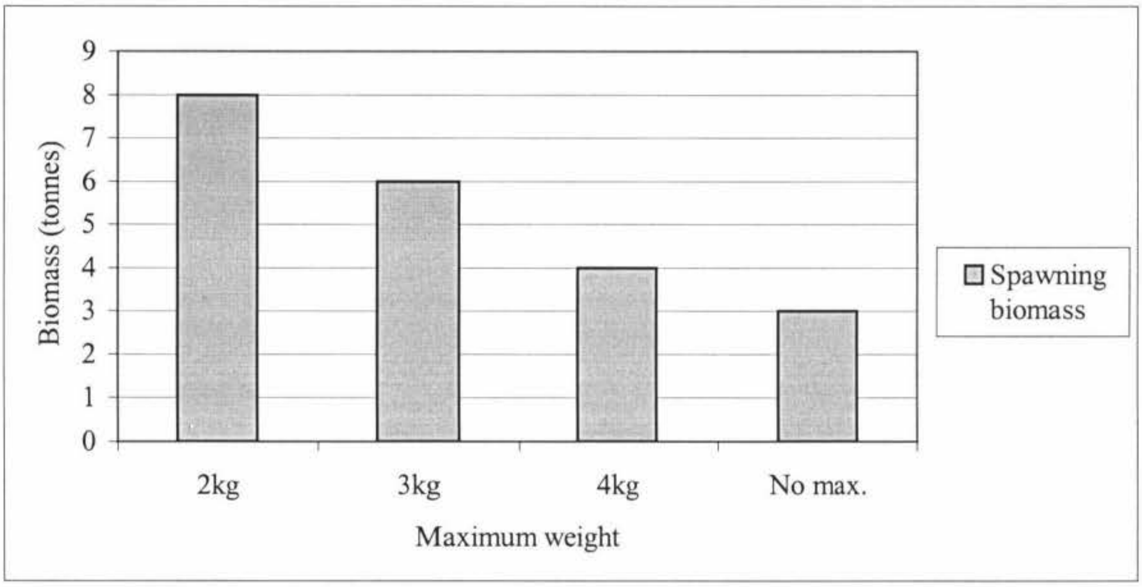

Figure 7.9. Levels of spawning biomass across varying maximum weight restrictions for unconstrained harvest and a minimum weight limit of 1.5kg.

This result is intuitive because a lower maximum legal weight allows sufficient time for year classes to recover to significant levels before spawning, after harvesting has ceased (Figure 7.10). This bears a high cost in that profits accruing to these policies will be comparatively lower, *ceteris paribus* (see Figure 7.8), because harvest is constrained through limiting the availability of a significant proportion of the most valuable

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[16] For further information, see the levels of spawning biomass for all analysed policies in Table A8.1 in Appendix 8.

cohorts. Although lower maximum weight limits provide for greater levels of spawning biomass (Figure 7.9) (and subsequently higher levels of recruitment), equilibrium stock levels are generally lower for those policies incorporating such a restriction. For example, the steady-state stock level at the lowest maximum weight limit (2kg) is up to 28 percent lower than that corresponding to no maximum weight restriction, *ceteris paribus*.[17] This reflects that after the terminal harvestable cohorts are heavily exploited (year classes 21 and 22 in Figure 7.10), in line with previous discussion regarding user cost, these severely exploited year classes must grow for up to nine years before spawning (Figure 7.10) and their low levels significantly weaken the total adult population. This effect is most pronounced when harvest is unconstrained, as quota constraints bind most noticeably on the cropping of the terminal harvestable year groups since these make up the majority of harvest.

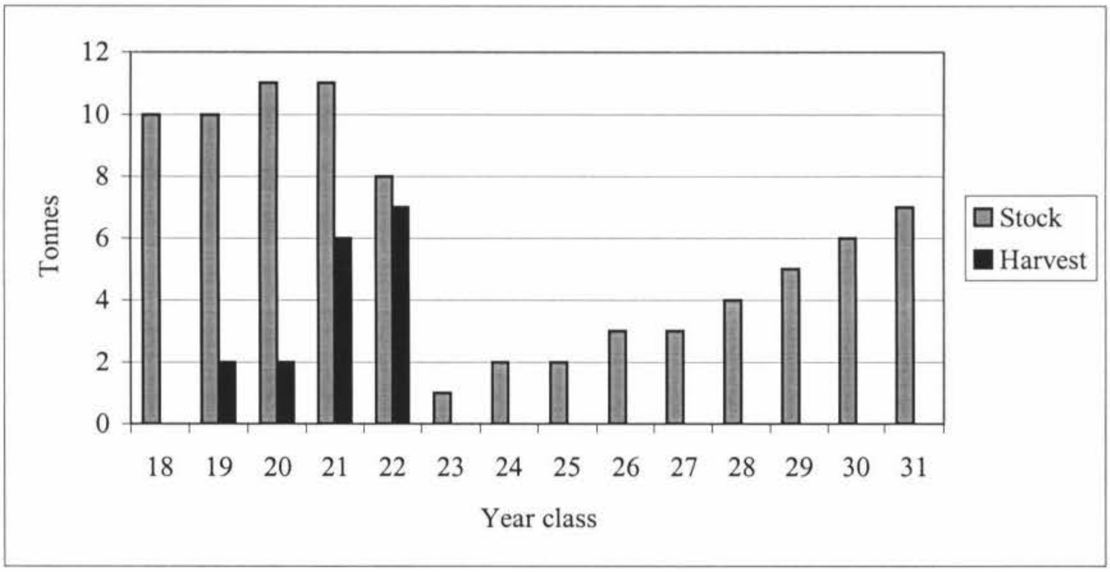

Figure 7.10. Equilibrium stock and harvest levels for the fourteen eldest year classes for a minimum weight limit of .5kg, a maximum weight restriction of 2kg, and no TAC.

Conversely, strong year classes are maintained across the majority of the most valuable cohorts with a higher or no maximum weight restriction (see Figure 7.6) and thus the total adult stock is larger for these policies. Although growth is promoted at low stock

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[17] For further information, see the equilibrium stock levels for all analysed policies in Table A9.1 in Appendix 9.

densities and recruitment increases (following the establishment of larger spawning cohorts) with the implementation of a lower maximum weight restriction, these are not sufficient to surpass the benefits accruing to the vulnerability of a high number of year classes to the fishery (see Figure 7.8).

Significant changes (an increase and decrease of 20 percent relative to current estimates) to the parameters defining the Beverton-Holt spawner-recruitment relationship have negligible effect on output, with the largest difference in NPV being 5 percent.[18] The small effect of a change in these parameters reflects the importance of physical growth, relative to recruitment, for the provision of net growth for harvest. This arises from the multiple-cohort nature of longfin populations, as the minor harvest of annual growth over a high number of year classes permits a reasonable level of exploitation. This minor harvest allows the continued cropping of cohorts until they reach the terminal harvestable age groups and are then cropped more heavily. In contrast, recruitment plays a lesser role, as annual growth has a cumulative effect on cohort biomass over time, due to the longevity of longfins, and therefore a significant adult population may establish even when elver return is low. For example, the equilibrium stock within the base model is 183 tonnes and recruitment at the corresponding level of steady-state spawning biomass (three tonnes) is just over 36kg.

It is intuitive that annual growth generally becomes more important, relative to biological recruitment, for the provision of surplus production that may be harvested sustainably from a multiple-cohort population as the number of *harvestable* year classes within this stock increases, *ceteris paribus*. At a broad level, this effect is observable in a comparison of these results for the longfin and the importance of recruitment to optimal harvest policies within Beverton-Holt frameworks incorporating species, such as prawns (*Penaeus* spp.), that have a low number of cohorts (Clark et al. 1973; Clark et al. 1979).

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[18] This robustness is particularly important because of the difficulty associated with the identification of these estimates and since the estimated relationship does not portray decreases in fecundity associated with increased exploitation (Jennings et al. 2001; Hoyle and Jellyman 2002).

The use of no maximum weight limit is important to the efficient management of the longfin fishery because this maximises the number of cohorts whose annual growth may be exploited during recovery and at equilibrium. However, the semelparity of longfins means that heavier levels of exploitation permitted within strategies that incorporate no maximum weight restriction significantly damage spawning biomass.

7.3.4 Importance of moderate exploitation rates

All quota constraints are binding in this model because it is more profitable in every case to maintain a higher rate of exploitation. This is illustrated in Figure 7.11, as NPV builds as restrictions on harvest are lifted.[19] This identifies the relative inefficiency of quota levels of 5 and 10 percent.

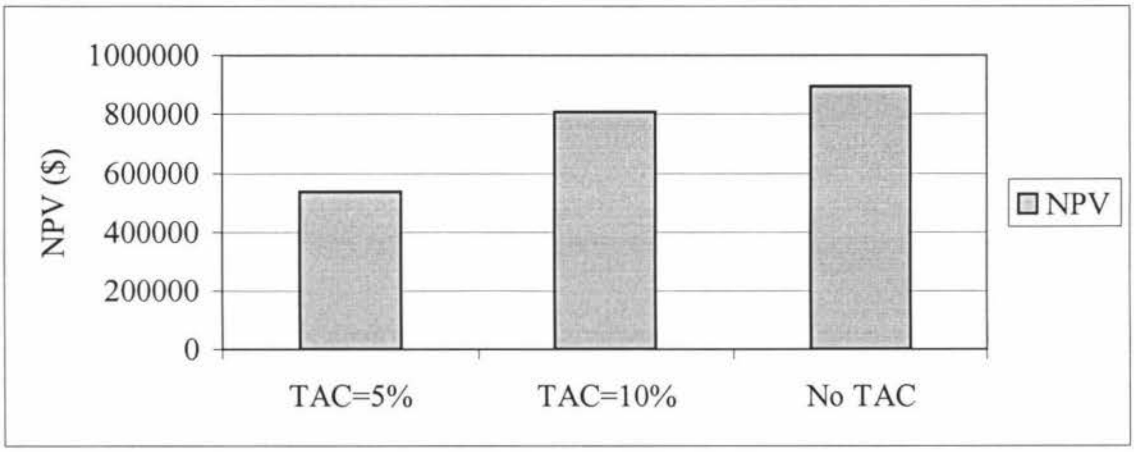

Figure 7.11. NPV across varying rates of exploitation for a minimum weight limit of 1.5kg and no maximum weight restriction.

The optimal exploitation rate consistent with the most efficient management policy (a minimum weight limit of 1.5kg and no maximum weight restriction) is 14.1 percent. This optimal rate is considerably higher than estimates of those that may maximise biological yield (5 to 8 percent) (Hoyle and Jellyman 2002). This demonstrates the increased optimism provided by the assumptions of this model, relative to those used

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[19] This effect is observable in more detail in the levels of NPV accruing to all alternative management strategies in Table A6.1 in Appendix 6.

by Hoyle and Jellyman (2002). First, deterministic recruitment reduces the frequency of extremely weak cohorts moving through the fishery. This effect is small given the lesser importance of recruitment relative to physical growth. Second, perfect selectivity reduces the cumulative impact of exploitation as only the most valuable year classes are exploited and the majority of these are only lightly fished. Third, density-dependent growth promotes harvest at low population densities.

While the optimal harvest rate is significantly higher than those estimates of Hoyle and Jellyman (2002), it is well below the 20 percent rate of historical exploitation reported by Jellyman et al. (2000). This demonstrates the general inability of longfin populations to maintain significant levels of exploitation, even with the incorporation of those favourable assumptions discussed above. This is a direct result of longfin biology, given slow growth (even when this is promoted at low stock levels) and the cumulative impact of exploitation promoted through the longevity of these eels.

The 20 percent rate of exploitation identified by Jellyman et al. (2000) is considered to be the maximum that would be encountered in this fishery. Although rates of current harvest in some regions may be close to those identified as optimal by model output, these are unsustainable since there is no incentive to maintain spawning biomass when property rights are poorly defined.

Exploitation significantly damages stock and spawning biomass (Figure 7.12)[20] as the rate of permitted exploitation increases. This reinforces the key role that semelparity plays in determining an optimal strategy. This finding also identifies the key role of physical growth since optimal management does not involve a large spawning biomass.

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[20] This is observable in greater detail in the equilibrium levels of spawning biomass and stock for all alternative management strategies. These are presented in Table A8.1 in Appendix 8 and Table A9.1 in Appendix 9 respectively.

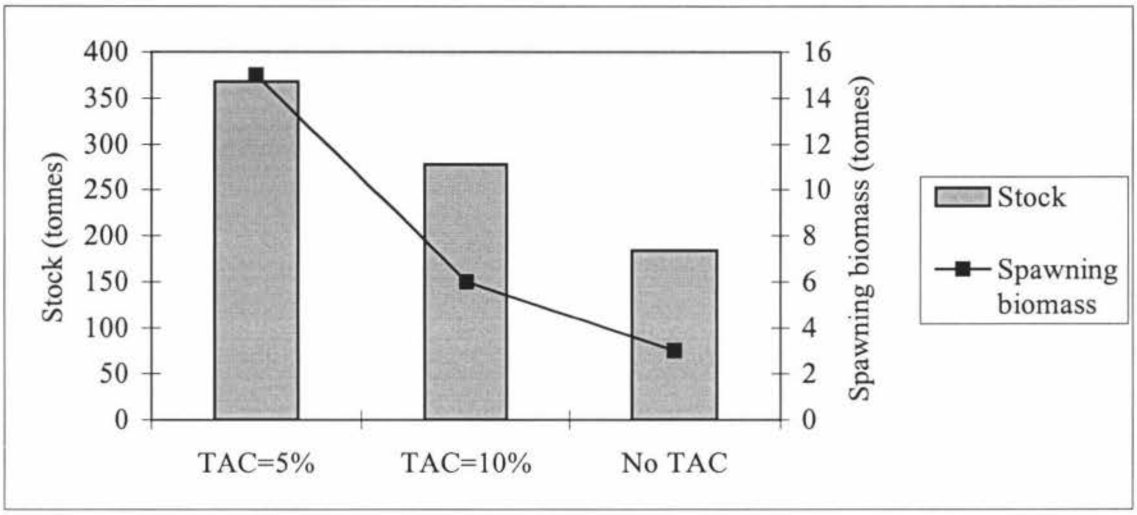

Figure 7.12. Stock and spawning biomass across varying rates of exploitation for a minimum weight limit of 1.5kg and no maximum weight restriction.

The slow growth of longfins impacts on the ability of fishers to maintain significant levels of exploitation, even in a framework incorporating deterministic recruitment, perfect selectivity, and density-dependent growth. Since harvested eels have never contributed to spawning, the maintenance of even a moderate rate of exploitation will severely damage future spawning cohorts and therefore recruitment. This can only be countered within this model through decreasing exploitation using a lower maximum weight restriction or harvest constraint, but these actions bear an efficiency cost.

7.3.5 The optimal recovery of longfin populations

All of the analysed models have the same nature of approach path for population and harvest as the base model. The recovery of stocks is a rather smooth increase towards equilibrium (Figure 7.13). Steady-state population levels decrease significantly as exploitation builds, as discussed in the previous section.

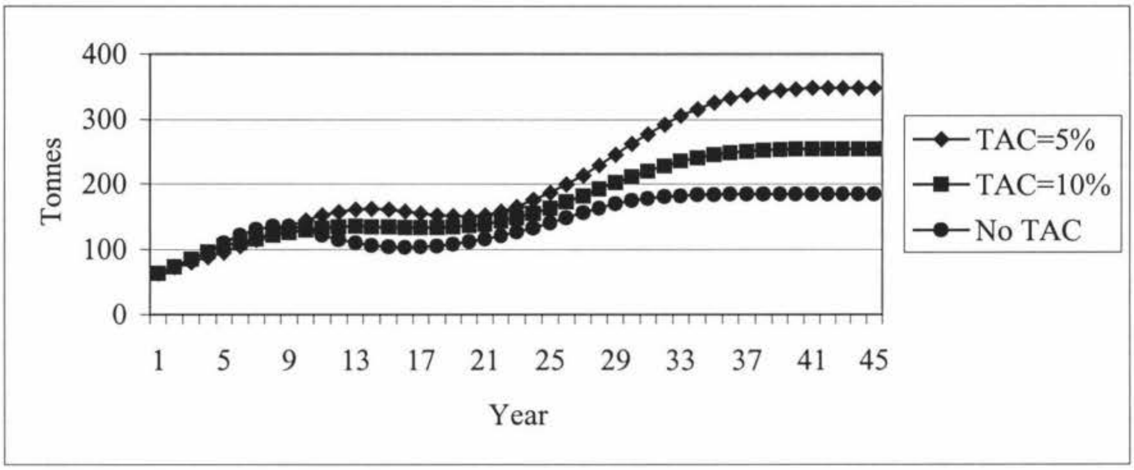

Figure 7.13. The recovery of stock levels across all TAC levels for a minimum weight limit of 1.5kg and no maximum weight restriction.

Total harvest experiences one significant fluctuation before resting at steady-state (Figure 7.14). This fluctuation represents the dynamics of adjustment as the sole owner bids to maximise the value of harvest by exploiting the most valuable cohorts, but also establish stable, significant levels of stock within each year class. This equilibrium takes considerable time to achieve given the delay between spawning and the entry of corresponding young into the fishery and the lengthy period it takes for the initial exploited cohorts to move through the population.

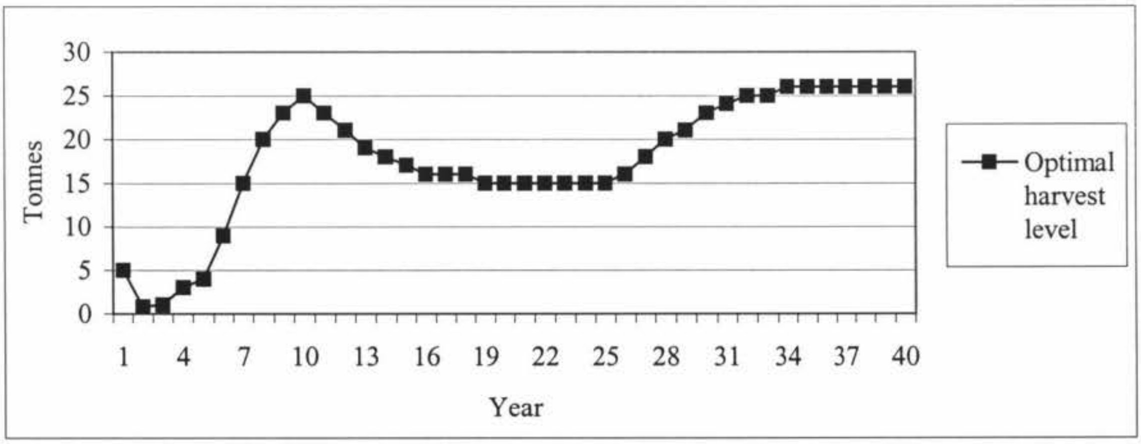

Figure 7.14. The time path of harvest for a minimum weight limit of 1.5kg, no maximum weight restriction, and no TAC.

Significant harvest may be supported rather quickly (Figure 7.14), given sufficient controls on initial exploitation. This optimistic, utilitarian view of recovery rests on an assumption of density-dependent growth, which has yet to be studied in significant detail, although a number of analyses support this proposition (see Section 2.4).

No periodic or pulsing behaviour is observed in the optimal harvest profile (Figure 7.14). The Most Rapid Approach Path (MRAP) is not optimal within this framework because the objective function after constraint-substitution is not additively separable in the state variables for the current period and that for the next year (Spence and Starrett 1975). Likewise, a bimodal harvest policy in its typical form, whereby younger reproductive cohorts with higher fecundity are partially exploited and older cohorts past some optimal breeding age are heavily fished (Walters 1969; Beddington and Taylor 1973; Reed 1980; Getz 1980), does not hold. This result is intuitive, given its reliance on the assumption that an individual within the species in question has more than a single reproductive contribution over its lifetime.

In contrast to these outcomes, the optimal age structure and harvest profile resemble the bimodal result of Moyle (1999), where the breeding cohort is protected and the more valuable non-reproductive year class is exploited. While the reproductive cohort is potentially vulnerable to exploitation in the analysis of Moyle (1999), the spawning year class is unavailable to fishers in the present study; therefore the protection/harvest of this age group cannot be endogenously determined. Nonetheless, the outcome is the same in that more valuable cohorts are exploited, but adequate stocks are retained so that these year classes may provide sufficient levels of spawning biomass to maintain harvest. This approach is intuitive for describing the optimal harvest of a long-lived, multiple-cohort, semelparous population.

Figure 7.14 demonstrates that either a moratorium or negligible levels of harvest would be required for a short period (around five years) at the initial stages of recovery to build spawning biomass. This ideally should be followed by some partial restriction on harvest to permit the stock to recover to more significant levels before heavier rates of exploitation are used. The QMS is particularly suited to the implementation of this transition management policy since TACs may be altered annually through manipulation of Annual Catch Entitlements (ACEs). The effectiveness of this policy would rely on the ability to formulate suitable stock assessment procedures, which is clearly problematic for longfins. This result therefore indicates that short moratoriums would be preferable to harvest restrictions for the recovery of longfin stocks.

7.3.6 Key factors underpinning efficient management of the longfin fishery

The characteristics of the most efficient management strategies demonstrate a number of important principles. First, user cost determines that harvest should be concentrated on the most valuable year classes, with exploitation ideally increasing as a year class approaches spawning given that the user cost accruing to its harvest is declining. Second, physical growth is more important to the provision of net production than recruitment as annual growth has a cumulative effect on cohort biomass. The significance of this and the user cost mentioned above are promoted through the longevity of longfins. Third, slow growth means that this fishery is unable to support a significant exploitation rate, even under the optimistic assumptions included within this model. Fourth, harvest and profitability increase with the exposure of a higher number of the most valuable year classes. This reflects the cropping of annual growth from a greater number of cohorts. Last, management strategies incorporating no maximum weight restrictions or higher levels of permitted exploitation place significant pressure on spawning biomass through increased harvest. In fact, semelparity underlies a significant inverse relationship (R^2=.59) (Figure 7.15) between efficiency and biological conservation, as measured through NPV and spawning biomass respectively, within the analysed policies.

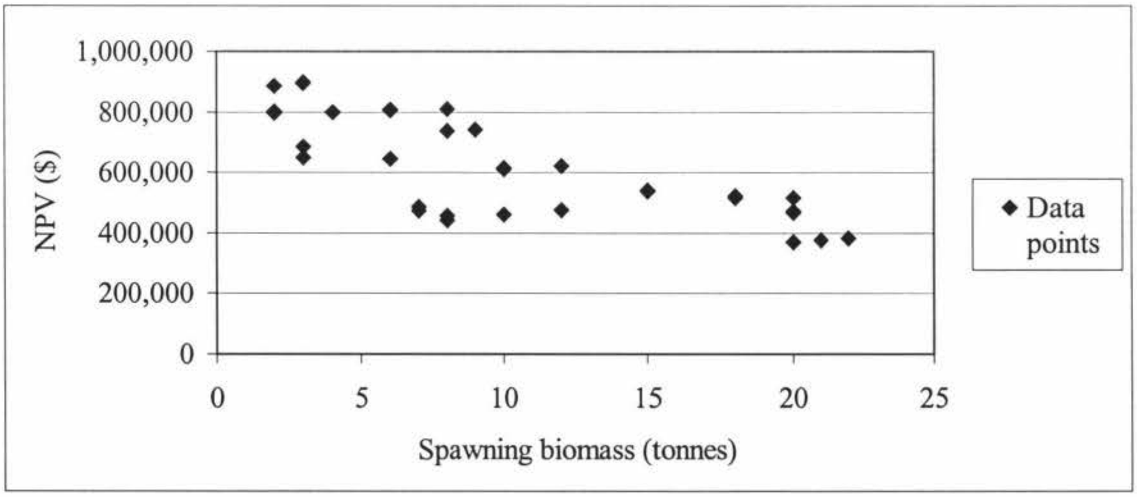

Figure 7.15. The inverse relationship between the profitability of the fishery and levels of spawning biomass.

7.4 Sensitivity analysis

The sensitivity of model output to changes within key relationships and parameters is particularly important because of the uncertainty involved in their estimation and specification. The results of this sensitivity analysis are interesting in that although absolute changes are observed, the optimality of the most efficient management policies is robust to the modification of parameter values and model relationships as the effects of these changes on model output are all relative. It follows that the results of sensitivity analysis are specified relative to the base model and alternative policies are not compared given that they remain sub-optimal. This provides for a concise presentation of the sensitivity of model output to parameter change without the loss of any important insight.

Previous discussion outlines that model output is significantly robust to large changes in price (Section 7.3.2) and the parameters defining the spawner-recruitment function (*a* and *b*) (Section 7.3.3). Additionally, changes in the cost parameter (a 25 percent increase and decrease relative to the estimated parameter) yield little response. In contrast to these results, the model is particularly sensitive to the strength of diminishing returns to the stock level, the discount rate, the density-dependent growth factor, and the specification of density-independent growth.

7.4.1 Sensitivity of model output to the strength of the stock-related cost externality

The base model is sensitive to the adjustment of the strength of the stock-related cost externality (i) within the cost function (Table 7.1). There is a positive relationship between a decreasing stock effect and NPV, which is counter-intuitive because this would be expected to increase costs and consequently lower profit. The optimal stock level falls with a reduction in i as there is decreased benefit accruing to the maintenance of a higher stock. The terminal harvestable year class is therefore cropped more heavily[21] as a smaller stock permits a lower level of equilibrium spawning biomass and

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[21] For example, the harvest of the terminal cohort increases by 40 percent, relative to the base model, when i=.25.

cost savings accrue to the heavier harvest of a single cohort, relative to the rest. The lower stock level also promotes growth, which permits a higher exploitation rate and consequently a greater level of NPV. This result indicates the importance of the growth effect once more.

Table 7.1. Changes in equilibrium population, spawning biomass, exploitation rate, and Net Present Value for alternative values of the stock-related cost externality, relative to the base model (i=1).

Strength of diminishing returns	Steady-state stock (% change)	Steady-state spawning biomass (% change)	Equilibrium rate of harvest (% change)	NPV (% change)
i=0	-13.1	-33.3	+11.3	+17.3
i=.25	-12.2	-30.6	+10.6	+15.7
i=.5	-11.1	-27.2	+9.1	+12.9
i=.75	-9.2	-13.7	+7.7	+8.1

It could be expected that heavier harvest of a younger cohort would be promoted, as the benefit accruing to this biomass, in terms of lowering future cost, would be decreased. However, this is dominated by the other components of the user cost accruing to the harvest of a young longfin year class.

7.4.2 Sensitivity of model output to the discount rate

The effects of alternative discount rates on model output are examined in relation to results for the base case (where δ=5 percent). NPV and the optimal equilibrium stock level fall significantly as the rate of discount increases, as bioeconomic theory suggests (Clark 1976a, 1985) (Table 7.2). Although the model is considerably sensitive to these alternative rates, wide deviation from the mean discount rate (5 percent) is not expected given a policy commitment to low and stable inflation in the 1989 Reserve Bank of New Zealand Act.

Table 7.2. Changes in the levels of equilibrium stock, harvest, and Net Present Value for alternative discount rates, relative to the base model (δ=5 percent).

Discount rate (%)	**Equilibrium stock (% change)**	**Equilibrium harvest (% change)**	**NPV (% change)**
δ=0	+33.3	+11.1	+929.6
δ=2.5	+16.4	+3.7	+269.7
δ=7.5	-16.4	-7.8	-54.7
δ=10	-33.3	-11.2	-76.4

Table 7.2 demonstrates that, in accordance with bioeconomic theory (Gordon 1954; Clark 1976a; Tietenberg 1996), the stock level at MSY (δ=0) is significantly higher than that for Maximum Economic Yield at existing weight restrictions, while equilibrium catch is also marginally greater. The model estimates an exploitation rate at MSY of around 12.3 percent, a moderate estimate only slightly greater than that identified by Francis and Jellyman (1999) (10 percent), but well above the rates reported by Hoyle and Jellyman (2002) (5 to 8 percent). This reinforces that longfin populations may support higher levels of harvest when growth is stimulated at low population density.

7.4.3 Sensitivity of model output to the density-dependent growth parameter

The base model is sensitive to changes in the density-dependent growth parameter (Table 7.3), with large deviations observable for harvest and consequently NPV. This reinforces the importance of physical growth to the conclusions of this model, as the bioeconomic principle in which slower/faster growth allows lower/higher equilibrium yield is demonstrated (Clark 1976a, 1985). 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 describing the spawner-recruitment relationship (Section 7.3.3).

Table 7.3. Changes in the level of equilibrium population, harvest, spawning biomass, exploitation rate, and Net Present Value for alternative values of the density-dependent growth parameter (ϕ), relative to the base model (ϕ=.0004).

Parameter change	Steady-state stock (% change)	Steady-state harvest (% change)	Steady-state spawning biomass (% change)	Steady-state rate of harvest (% change)	NPV (% change)
ϕ=.00036 (10% decrease)	-10.9	-15.4	N/C	-5.6	-17.8
ϕ=.00038 (5% decrease)	-4.9	-7.7	N/C	-3.5	-8.1
ϕ=.00042 (5% increase)	+3.8	+15.4	+33.3	+10.5	+12.5
ϕ=.00044 (10% increase)	+7.7	+19.2	+33.3	+9.8	+23.3

The optimal age structure of harvest is robust to the modification of the density-dependent growth factor, as changes to this parameter affect all year classes at the same rate since it is constant across all age groups in equilibrium. This can be understood in terms of increases to this parameter having an absolute effect, which promotes total harvest, but not a relative effect, as user cost (and therefore the proportion of each year class within the optimal catch) does not change given the constant growth rate of all adults in steady-state. An additional factor of interest is that even when the growth factor is 10 percent higher than that estimated, the optimal exploitation rate (15.7 percent) remains below an estimate of historical harvest identified by Jellyman et al. (2000) (20 percent). This reinforces the sub-optimality of current harvest.

7.4.4 Sensitivity of model output to density-independent growth

The same method used to calculate growth rates for juvenile cohorts (Section 6.5) is used to calculate fixed growth factors for adult year classes to test the sensitivity of the model to the incorporation of a density-independent specification. The addition of fixed growth factors leads to a form of periodic cropping (Figure 7.16), whereby harvest is maintained at periodically low levels to allow for the rebuilding of stocks.

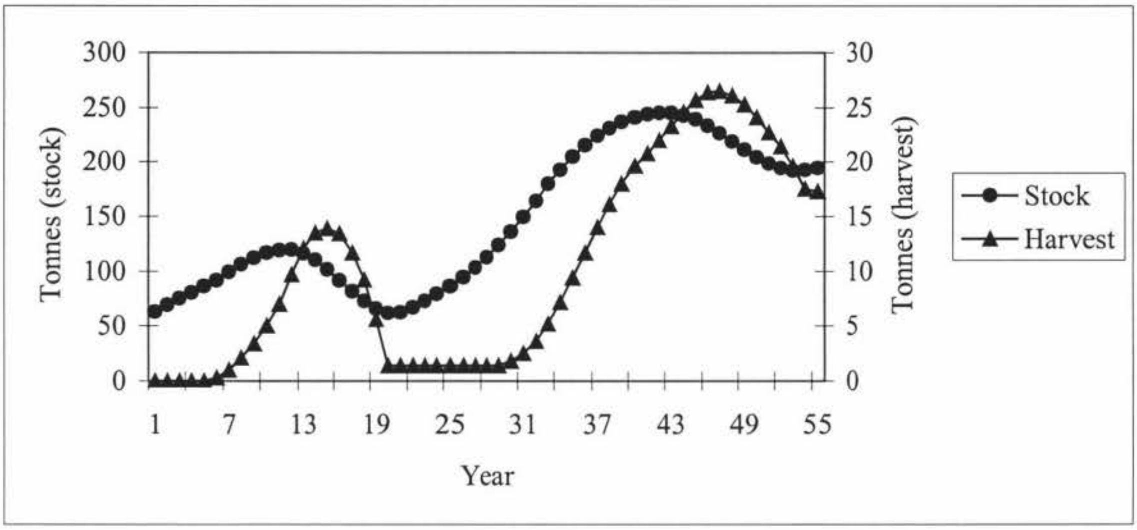

Figure 7.16. Stock and harvest profiles when growth is density-independent.

Harvest builds gradually to a peak, but as stocks diminish this must then decline to a periodic low once more to allow the population to recuperate. This demonstrates the inability of longfin populations to recover quickly to a point where constant exploitation may be sustained when growth is density-independent. This harvest profile has a large effect on total catch, which halves over the first fifty years. NPV is consequently lower, falling by around 57 percent over the planning horizon. This reflects the importance of density-dependent growth in the provision of surplus production for annual harvest within this model.

Exploitation moves away from the terminal harvestable cohorts towards younger age groups within the most valuable year classes when growth is density-independent, since the growth rate of adults is decreasing in age. This decreases the user cost accruing to the cropping of younger cohorts, as the benefit that corresponds to the future growth of a year group is reduced.

7.4.5 The incorporation of stochastic recruitment

The portrayal of recruitment as a deterministic process within this model suggests that those policies incorporating lower levels of spawning biomass may become less valuable with the inclusion of stochasticity in the spawner-recruitment relationship. This may be incorporated through the multiplication of the spawner-recruitment relationship by a white noise variable with a known probability distribution (Reed

1979). This is left as a topic for further research given the present lack of information pertaining to the probability distribution of longfin recruitment. This is also consistent with the minor effect that the addition of stochasticity has on optimal policies (Ludwig and Walters 1982; Clark 1985; Conrad and Clark 1987; Benton and Grant 1996), especially since the model is robust to significant changes in the parameters defining the spawner-recruitment relationship.

The use of a single spawning class is also unsuitable for investigating the effects of stochasticity, as wide variability in recruitment and low initial stock sizes would lead to no recruitment in some years. These effects would compound as these year classes subsequently fail to breed. This outcome would significantly skew results, as maturity over a range of ages would help to buffer such extreme outcomes in reality. This identifies that the incorporation of stochastic recruitment in this model would ideally require the portrayal of sexual maturity over a number of age groups. Such an addition is likely to be nontrivial.

7.4.6 Summary

The sole harvest of the most valuable year classes is particularly robust to all changes to original parameter values and key relationships. Moreover, the heavy exploitation of the terminal harvestable year classes only becomes less favourable when adult growth rates are decreasing in age. These results indicate that the multifaceted nature of user cost accruing to this long-lived species prevents a change in one factor from significantly modifying the most efficient weight restrictions. In contrast, sustainable levels of harvest experience some variation. Although these are not large when growth is density-dependent, the ability of longfin populations to sustain constant exploitation is questionable when growth rates are fixed and decreasing in age. These factors demonstrate that while recommendations for higher minimum weight restrictions and no maximum weight limit are robust to parameter modification, further research into density-dependent growth is required before an optimal exploitation rate is identified.

7.5 Efficient management of the longfin fishery

The objectives of fishery managers are numerous, but it has been the primary focus of the above discussion to identify the implications of model output for the efficient management of longfin populations. This follows the biological focus of previous models (Jellyman et al. 2000; Hoyle and Jellyman 2002) and the basis of this work in bioeconomics. The efficient utilisation of the longfin resource requires the harvest of the most valuable year classes, leaving the minimum required to spawn and maintain recruitment. While maximising efficiency may be a commendable objective in theory, the practicality of maintaining this balance is questionable because of the level of abstraction contained within the model.

In reality, uncertainty presents a significant barrier to the implementation of the optimal strategy. Efficient management requires the identification of appropriate TACs in order to reduce competition among fishers and permit sufficient numbers of eels to reach spawning. Accurate estimates of sustainable catch are of increased importance in this fishery as commercial harvesters have a price incentive to exploit animals approaching sexual maturity. Yet a lack of stock information prevents the estimation of suitable TACs and biological reference points. Longfin biology, for example territoriality, also hampers the effective construction of stock assessment techniques. Significant heterogeneity among regional stocks in regard to growth, species composition, and population density also hinders the identification of sustainable harvest levels. Limiting exploitation to the point where sufficient numbers of individuals reach sexual maturity is also complicated by uncertainty surrounding levels of escapement and spawner-recruitment relationships, both of which are impossible to identify with certainty because of longfin biology and environmental variability. Moreover, this makes it difficult to identify the implications of a decrease in fecundity arising from increased exploitation. These factors indicate that extensive application of the optimal policy would fail to protect these populations. In fact, the broad implementation of any harvest-based management system is unlikely to safeguard longfins, given that spawning biomass is sensitive to exploitation and uncertainty prevents the estimation of appropriate TACs.

Efficient management has a number of costs that are not incorporated within this model. First, concentrating harvest on the largest animals will incur significant costs in terms of decreased catch for both recreational and cultural fishers, especially since both user groups prefer larger longfins to any other type of freshwater eel. Second, the pressure that efficient management places on spawning populations, and consequently recruitment, threatens the continued provision of existence value and ecological stability in many freshwater ecosystems. These factors may be thought of as additional components of user cost that are not included in the model, but would increase the importance of the need to balance the higher value of larger longfins to society and the future spawning value of these eels.

Balancing utilisation and conservation for a slow-growing species with no extensive stock information is a difficult undertaking. This is exacerbated through multiple user groups and semelparity, as the latter underlies an intuitive inverse relationship between exploitation and spawning biomass. These difficulties identify the importance of a precautionary approach to management (Hoyle and Jellyman 2002).

7.6 Construction of a suitable management strategy

Hoyle and Jellyman (2002) provided a detailed analysis of alternative management strategies for longfin fisheries. These authors proposed that low exploitation rates (5 to 8 percent) should be used to maximise yield in areas open to fishing. Additionally, Jellyman et al. (2000) and Hoyle and Jellyman (2002) reported that maximum weight limits of 2kg or lower were required to significantly increase spawning biomass. This model demonstrates that these rates of harvest and low maximum weight restrictions have severe negative implications for economic efficiency. The economic costs accruing to the use of area closure (Jellyman et al. 2000; Hoyle and Jellyman 2002) to protect spawning populations are also obvious (Section 3.6.6). The relationship between these factors and the optimal policy within this model can be summarised in a continuum representing the divergence between those policies that offer significant protection for spawning biomass and those that maximise efficiency (Figure 7.17).

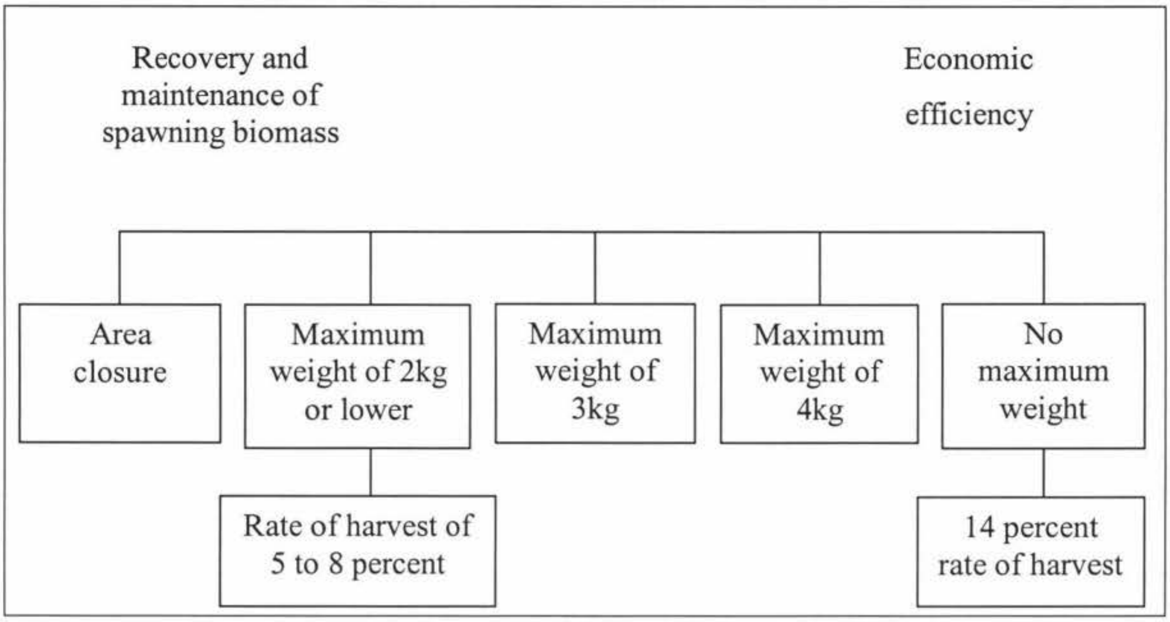

Figure 7.17. The divergence between policies designed to maintain robust longfin populations and those that maximise economic efficiency.

This continuum and discussion in Section 7.5 suggest that the application of an ITQ system, with no maximum weight restriction, across the entire fishery would be insufficient to halt the decline of longfin populations. ITQ management relies centrally on the identification of a TAC consistent with both conservation and economic efficiency; however, longfin biology, in particular slow growth, longevity, and semelparity, means that surplus production within these populations is too low to maintain significant levels of exploitation, while also sustaining a robust spawning biomass. This is demonstrated in model output, as even under favourable assumptions only a moderate rate of equilibrium harvest is maintained. Consequently, harvesters must significantly lower the strength of those cohorts approaching spawning in order to maintain moderate rates of harvest.[22] This prevents the identification of an optimal TAC that may satisfy efficiency and conservation goals. Even if a suitable catch limit were identified, the protection offered by a harvest-based regulatory system is negated through uncertainty and the high sensitivity of longfins to exploitation (Section 7.5).

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[22] The true profitability of optimal harvest levels is difficult to assess, however, as the NPV figures identified in this analysis (Table A6.1 in Appendix 6) are clearly not representative of reality given the strength of the assumptions used in the model and poor information.

The insufficiency of using past harvest to calculate TACs is also evident. While existing harvest is too high to permit the recovery of these populations (Section 7.3.4), this data portrays a false sense of stability because of poor species composition data, the historical effects of harvest on spawning biomass (Jellyman et al. 2000), high annual variability in catch and effort statistics (Jellyman 1993), and the difficulty involved in observing recruitment trends.

In conclusion, while New Zealand has an established ITQ management system in place, the use of any harvest-based regulatory policy, without significant investment in area closure, will therefore not protect longfin populations through the recovery and maintenance of spawning biomass. This suggests that the implementation of a management policy that lies somewhere on the continuum between the most efficient policies and those that sufficiently safeguard spawning biomass will provide for neither of these goals, as it is obvious that this would involve harvest restrictions (Figure 7.17). A maximum weight limit could be used to compensate for a lack of protection for breeding populations offered by harvest restrictions. This, too, offers little protection since harvesters have a strong economic incentive to harvest eels before they reach this limit, other user groups may capture eels above this weight, commercial fishers may exploit protected animals illegally, and discarding larger individuals may cause significant mortality, given their high catchability.

These factors indicate that to satisfy the divergent goals illustrated in Figure 7.17, a suitable management strategy would encompass two important components, permanent area closures to rebuild/maintain breeding populations and managing areas open to fishing (henceforth open areas) to maximise efficiency.

7.7 Area closures

An extensive discussion of the benefits and limitations of area closure is presented in Section 3.6.6; therefore this section focuses on the key factors that must be considered in the use of area closures in this integrated management strategy. Area closures hold particular promise for rebuilding and maintaining spawning biomass within longfin populations (Chisnall and Hicks 1993; Jellyman 1993; Jellyman et al. 2000; Hoyle and Jellyman 2002), given significant uncertainty (Walters 2000), the limited home ranges of longfins, and high larval spillover (Sladek Nowlis 2000; Hoyle and Jellyman 2002).

Current area closures are limited in the provision of these benefits (Jellyman 1993) due to impeded in- and out-migration, illegal fishing, slow growth, and low population densities.

To overcome these problems, it is important that the size of areas permanently closed to all types of eel fishing is significantly increased. These also need to be established in more productive regions, for example in hydroelectric lakes[23] where animals may reach maturity up to four times faster than those in forested streams due to low water velocity, low densities (which would obviously change if these areas were seeded too heavily with elvers or juveniles), high water temperatures, and food abundance (Chisnall and Hicks 1993).

The level to which migrating sexually mature eels are exploited is not clear, as while a number of authors report that these eels are seldom exploited (Annala et al. 2001; Hoyle and Jellyman 2002), various authors stress the need for a maximum weight restriction to protect them (Chisnall and Hicks 1993; Jellyman 1993; Chisnall et al. 2002). If the latter is true, the inefficiency of a maximum weight restriction (Figure 7.17) reflects the costs that may accrue to the implementation of this policy in open fisheries if used to protect spawners migrating from closed areas. Variability in the month of migration for mature animals of both genders also does not permit the restriction of this limit to a short period within the fishing season. It may therefore be more productive to form closed waterways, instead of localised closures within a waterway.

The use of closed waterways would avoid the need for a maximum weight limit, overcome problems associated with the diffusion of eels from closed areas to exploited stocks, and protect both males and females, given that the strength of each gender within a population differs between coastal and inland areas. For example, the closure of the Manawatu River and its tributaries would provide a significant weight of annual migrants if populations here were permitted to recover. The suitability of this waterway

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[23] Although successful migration does pose some obvious problems, a significant amount of research is under way to overcome the problems posed by barriers to the movement of both elvers and sexually mature eels (Jellyman et al. 2000).

is aided in that it is of significant size (it is 170km in length), no hydroelectric dams are present that would hamper the migration of eels, it contains a significant proportion of longfin eels relative to shortfins (Beentjes and Chisnall 1998), and the lower reaches are ideal for promoting the growth of longfins given their passage through pastoral land (Chisnall and Hicks 1993). Seeding with elvers and juveniles may also help to rebuild populations (De Leo and Gatto 2001). However, monitoring would be required to minimise overstocking and subsequent losses in growth rate and the metamorphosis of females to males at high densities.

There are a number of limitations to the use of area closures. The identification of suitable areas of sufficient size may be difficult because of heterogeneous growth and population density between localities and since little is known of the reproductive biology of the longfin. Illegal fishing and the misreporting of catch may also be difficult to prevent as harvesters have a short-term incentive to crop closed areas. Enforcement costs may therefore be high as non-compliance is difficult to observe.

Area closures will also affect the provision of fishing opportunities among users and regions through the displacement of existing commercial, customary, and recreational fisheries in productive waterways, especially since a significant area will have to be closed if populations are to be sufficiently protected (Jellyman 1993; Jellyman et al. 2000). The extensive use of closed areas will therefore have significant economic and social costs as these lead to decreased fishing among all user groups. Area closures will also lower the profitability of the fishery through preventing the cropping of shortfins from these waterways, as gear cannot distinguish between species, post-capture removal is inefficient, discard mortality would decrease the value of closed waterways for protecting longfin populations, and fishers would have an economic incentive to retain longfins. It is also important to weigh together the relative benefits of closed waterways versus closed areas within waterways. While the former will maximise the benefits of area closure through minimising the exposure of individuals to harvest during migration or after diffusion, it may incur greater distributional effects. These distributional effects may promote the need for mediation and compensation.

Initial decreases in profits are a general result of fishery management aimed at reducing overexploitation (Clark 1985) and these would be smaller with area closure than with the collapse of the entire fishery. This identifies the importance of bearing current costs to build a resource of greater future worth, especially since a number of important non-market values accrue to this species and harvest-based regulation offers little protection for spawning biomass.

7.8 Managing open areas to maximise efficiency

Successfully managing open areas would require three important conditions to be met:

1. the sustainability of recruitment, which depends on a sufficient weight of females reaching sexual maturity annually in closed areas
2. the identification of efficient management policies, and
3. the allocation of property rights among multiple user-groups to maximise the value of these policies.

The first component relies on those factors discussed in Section 7.7. The second and third factors require further examination. In contrast to managing open areas conservatively to maximise yield (Hoyle and Jellyman 2002), this model demonstrates that significant gains above those accruing to such policies may be earned if economic efficiency is the operational goal of fishery managers. The optimal strategy to efficiently exploit longfin populations in these areas would ideally encompass those characteristics identified by the model: a high minimum weight limit, no maximum weight restriction, and a moderate exploitation rate.

7.8.1 Suitability of recommended weight restrictions

A higher minimum weight restriction would be relatively easily to implement, given harvester preference (see Question 16A in Appendix 3) and the use of escape tubes and mesh sizes consistent with this greater size. An optimal minimum weight limit for the entire fishery would be lower than that identified by model output (1.5kg). This follows the reduction in yield, and consequently profit, that would accompany the introduction of this restriction (a minimum weight limit of 1.5kg) in habitats containing both species, since the majority of shortfins spawn beneath this weight (Jellyman 1993) (see Section 7.3.1). Nonetheless, this optimal minimum weight restriction is of importance as it demonstrates that individuals in a multiple-cohort semelparous population should ideally be exploited more heavily as they approach spawning age because the user cost accruing to their harvest declines. Since the shortfin has similar biological characteristics to the longfin (that is, it is also semelparous and grows for many years before sexual maturity), it is apparent that fishing populations of both species at a higher minimum weight restriction will have significant benefit for maximising the value of this fishery and these gains will be increasing in the level of minimum weight imposed. The identification of an optimal minimum weight for fisheries that contain both species is an important area for further research.

The use of no maximum weight restriction is efficient and overcomes the limitations associated with the implementation of a maximum weight limit discussed in Section 7.6.

7.8.2 Management of exploitation in open areas

The efficient management of open areas also requires the allocation of fishing rights among multiple user groups and the calculation of an appropriate exploitation rate. Although competition for fish, the territorial nature of longfins, and high larval spillover suggest that area quotas may be a suitable method for allocating such rights, the heterogeneous nature of these stocks, diffusion, and the migratory patterns of elvers and spawners present large obstacles to their effective establishment. Therefore while the application of an ITQ system to the *entire* fishery would be unsustainable, it is suggested that this may be suitable for the management of open areas within this integrated management strategy. This would allow the use of a well-established system with a clearly defined process that is bound in legislation (Te Waka a Maui me ona Toka Mahi Tuna 1996) and theoretically solve the property rights problem present in this fishery.

Uncertainty and longfin biology would still prevent the objective calculation of TACs using traditional stock assessment methods (Ricker 1975; Quinn and Deriso 1999). It is proposed that the identification of suitable TACs would be aided within this integrated management strategy through a number of important factors that would provide a buffer to the detrimental effects of overestimating sustainable harvest. First, density-dependent growth would assist the recovery of populations if harvest were too high. Second, recruitment from spawners reaching sexual maturity in closed populations would help to safeguard the national longfin stock from recruitment overfishing during the investigation of sustainable exploitation rates. The effectiveness of the buffer provided by these factors would depend on the strength of density-dependent growth and a conservative estimate of the total size of area closures initially required. With the security provided by this buffer, fishery managers could use active adaptive management to further their knowledge of sustainable rates of exploitation, the size of area closures required, and feasible levels of allocation among the alternative user groups present in the fishery. Active adaptive management is accelerating the generation of information for a fishery system through deliberately "probing" this system (Walters 1986; Charles 2001). This strategy could encompass the calculation of a precautionary risk-averse initial state, incorporating overly conservative estimates of the size of area closures and TACs required, and then involve the monitoring of the effects of gradually releasing these constraints. Monitoring is of primary importance to the successful implementation of this system and therefore the development of appropriate methodologies is an area in which further research is recommended.

The incremental movement towards the identification of appropriate exploitation rates using active adaptive management may be problematic to implement because of user group pressure, environmental variation, and the longevity of longfins, which would increase the time required to identify a sustainable management strategy. These limitations may be significant, but an alternative approach is not easily identified given the current overexploited state of these stocks, longfin biology, and poor information. This may change with the collection of better data, for example for catch and effort, as the strength of populations may become more apparent with the development of stock assessment procedures. Nonetheless, there are obvious barriers to forming accurate estimates of population size for this species, indicating that an active adaptive management approach may be the only viable means of working towards the identification of sustainable harvest levels.

The optimal rate suggested by model output is likely to overstate sustainable harvest due to assumptions of perfect selectivity and density-dependent growth. Nonetheless, optimal harvest rates in open areas will be promoted in that stocks will not require the maintenance of spawning biomass because of the provision of spawners from closed areas. This would help to overcome some of the losses accruing to area closure and avoid the need for a transition management policy incorporating a short moratorium during the initial stages of recovery, which is necessary during the optimal recovery of stocks to increase the number of eels reaching sexual maturity (Section 7.3.5). A multiple species analysis and the validation and measurement of density-dependent growth are required before any recommendations for an optimal rate for all open areas can be formed. However, limitations facing stock assessment and the sensitivity of the optimal exploitation rate to changes in model parameters, which are far from certain, indicate that greater gains may be made through active adaptive management.

The use of ITQ management in open areas would also be promoted through the removal of incentives for harvesting larger eels. Higher minimum weight restrictions will reduce the motivation to high-grade quotas, but unless a minimum weight limit is set at 1.5kg - and it is obvious that this would incur significant cost through lowering shortfin yield - some competition for larger eels would remain. It is difficult to identify an efficient solution to this problem.

Competition for profitable unexploited fisheries where CPUE is higher will weaken as these become less common, unless, of course, prices significantly increase or a new fishing technology is developed. Unexploited fisheries will also decline with area closure. Additionally, although all user groups may compete for more accessible fisheries in order to lower travel costs, low CPUE and the use of adaptive management to identify more effective TACs would help to limit this effect across the fishery as a whole.

8. SUMMARY AND CONCLUSIONS

8.1 Background

The longfin eel is of importance to New Zealand society. The commercial fishery for freshwater eels provides significant levels of export revenue and employment in both this industry and through multiplier effects. This fish is also an endemic species and thus is a valuable component of New Zealand's biodiversity. The longfin is also ecologically valuable as a top predator in many freshwater ecosystems and supports two important non-market fisheries: recreational and customary.

Longfin recruitment has declined by 75 percent since commercial harvest reached significant levels in the early 1970s (Jellyman et al. 2000). This has occurred for a number of reasons. First, the longevity and high catchability of longfins means that the cumulative impact of even low levels of exploitation can have extremely deleterious effects on spawning biomass (Jellyman et al. 2000; Hoyle and Jellyman 2002). Second, all fishing mortality always occurs before spawning because of semelparity. Third, environmental variability, poor catch and effort data, territoriality, catadromy, and panmixis render stock size, annual fishing mortality, and recruitment unknown. This complicates the formation of effective management strategies. Fourth, fishers within a number of user groups compete for larger longfins, especially in the commercial fishery, where price increases with the size of individual eels. Fifth, the commercial exploitation of longfins is promoted through low harvesting costs and efficient gear. Last, historical regulation has failed to protect spawning biomass. This is likely to continue because even though the South Island fishery has been under ITQ management since October 2000, and the North Island will follow in 2003, a lack of information has stimulated the use of past harvest levels, consistent with overexploitation, to calculate TACs. These factors have all contributed to the severe recruitment overfishing of longfin populations.

This overfishing indicates the need for the identification of suitable regulatory systems for the recovery and maintenance of longfin stocks. The primary objective of this research is therefore to investigate sustainable management strategies for the New Zealand longfin eel fishery while considering its biological and economic characteristics. This investigation is based on the development of a deterministic multiple-cohort bioeconomic model that is used to gain insight into the nature of efficient harvesting strategies and identify the relative optimality of alternative regulatory policies. This analysis extends existing literature through analysing this fishery from a bioeconomic perspective.

The model used in this research is based on the age-structured dynamic optimisation model of Conrad (1982). This work is extended to include density-dependent relationships, the analysis of approach paths, and the portrayal of the complex biology of the longfin eel. While partial solution of the analytical model provides some insight into the optimal harvesting strategy, the complexity of the problem motivates the use of numerical non-linear programming, utilising the GAMS/CONOPT2 solver, to investigate this problem to the level of detail required. While this involves truncation of the model to finite time, this approach allows for an analysis of the relative efficiency of alternative management strategies.

8.2 Key results

This research provides interesting insight into the formulation of suitable regulatory policies for the longfin fishery.

First, the use of past harvest to calculate TACs is entirely inappropriate. Model output indicates that without permitting recovery, the perpetuation of unsustainable exploitation through TACs will be inefficient because yield will be sustained below higher, more profitable levels and fishers will face increased costs as eels become more scarce. The collapse of populations becomes more likely, as spawning biomass is not permitted to recover to more robust levels. The use of past harvest data to calculate TACs is also unsustainable due to poor species composition data, the historical effects of harvest on spawning biomass (Jellyman et al. 2000), high annual variability in catch and effort statistics (Jellyman 1993), and the difficulty involved in observing recruitment trends.

Second, effective allocation of property rights is required if harvesters are to have an incentive for conservation. Simulation of open access conditions intuitively indicates that competition for fish among commercial fishers promotes overexploitation. This effect will be magnified in reality through the presence of recreational and cultural fisheries that are not represented in the model. These factors indicate the importance of allocating property rights in these fisheries.

Third, harvest should be concentrated on the most valuable year classes, with exploitation ideally increasing as a year class approaches spawning because the user cost accruing to its removal is declining. This decreasing user cost is evident in that the benefits for maintaining year class strength, in terms of future growth, movement into more valuable year classes, and lowering harvesting costs (through a stock-related cost effect), decline as an eel approaches the time that it is no longer vulnerable to the fishery. These effects play a pivotal role in determining an optimal age structure of harvest for longfins because of the longevity and semelparity of this species. This optimal age structure is particularly robust to parameter change, including a price that is constant across all year classes. Additionally, exploitation remains concentrated on the most valuable year classes when the cohort growth rate is density-independent and decreasing in age. These results indicate that a higher minimum weight restriction than that used under the existing regulatory system will increase efficiency, with these gains increasing in the level of minimum weight imposed.

Fourth, physical growth is more important than recruitment to the provision of net production as annual growth has a cumulative effect on cohort biomass. The use of no maximum weight limit therefore provides for greater harvest and profit through exposing the surplus production of a higher number of the most valuable year classes to the fishery. This effect is promoted when growth is stimulated at low stock densities. This is demonstrated in that the fishery cannot rapidly recover to the point where constant exploitation may be maintained when growth is density-independent.

Last, the optimal exploitation rate consistent with the most efficient management policy (a minimum weight limit of 1.5kg and no maximum weight restriction) is 14.1 percent. This optimal rate is considerably higher than estimates of those that may maximise biological yield (5 to 8 percent) (Hoyle and Jellyman 2002). This demonstrates the increased optimism provided by the assumptions of this model relative to those used by these authors. The most important are density-dependent growth and perfect selectivity. The optimal rate identified by model output is well below the 20 percent rate of historical exploitation reported by Jellyman et al. (2000). This demonstrates the general inability of longfin populations to maintain significant levels of exploitation, even with the incorporation of these favourable assumptions. This is a direct result of longfin biology, given slow growth (even when this is promoted at low stock levels) and the cumulative impact of exploitation.

Slow growth means that even moderate rates of exploitation within the most efficient management policies significantly damage spawning populations. This is promoted when no maximum weight restriction is in place because heavily exploited terminal fishable cohorts have little time to recover before spawning. While low levels of consequent recruitment lead to lower population levels, and thus increased growth, semelparity underlies a trade-off between efficiency and biological conservation, as measured through NPV and spawning biomass respectively, within optimal policies. This is intuitive because any harvested eel has always yet to spawn.

8.3 Policy implications for the management of the longfin fishery

Managing the fishery to maximise efficiency will place significant pressure on spawning biomass, given the inverse relationship between these factors. In fact, spawning biomass is so low within the most efficient policies that populations are likely to collapse if these were extensively applied. Competition for larger longfins between fishers in a number of user groups indicates the need for an accurate specification of harvest limits in order for a sufficient number of individuals to reach spawning in order to maintain longfin populations. This is extremely complicated due to uncertainty, as a lack of stock information prevents the estimation of an appropriate TAC. This is hampered further by regional heterogeneity. Moreover, uncertainty surrounding the reproductive biology of longfins, especially the relationship between spawning and recruitment, complicates the identification of an efficient balance between the number of individuals reaching spawning and the exploitation of large longfins. The risk of collapse posed by harvest-based regulation and uncertainty is compounded when the sensitivity of larger animals to exploitation is considered. These factors suggest that the use of any harvest-based regulatory system without significant investment in area closure will therefore fail to protect longfin populations through the recovery and maintenance of spawning biomass.

A lower maximum weight limit is also unlikely to be useful in increasing breeding populations. Harvesters have a strong economic incentive to harvest eels before they reach this limit and other user groups may exploit eels above this weight, either legally or illegally, depending on whether the limit applies to all user groups. Additionally, discarding larger animals may cause significant mortality, especially given their frequent capture because of high catchability.

These factors and the divergence between those policies that protect spawning biomass and maximise efficiency suggest that the most suitable management strategy would encompass two important components, permanent area closures to rebuild/maintain breeding populations and managing open areas to maximise efficiency.

Existing longfin populations that are closed to fishing are insufficient to safeguard this species (Chisnall and Hicks 1993; Jellyman 1993; Hoyle and Jellyman 2002) due to their small size and their slow growth. Additionally, both in- and out-migration are frequently impeded by hydroelectric development and natural factors. It is therefore important to form area closures of significant size in productive habitats. Area closures are the most effective means of maintaining significant levels of spawning biomass given uncertainty, longfin biology, and the inefficiency and operational difficulties associated with lower maximum weight restrictions. The use of area closure would also overcome the need for a short moratorium to establish increased levels of spawning biomass before higher levels of exploitation may be sustained. The use of closed waterways would be particularly effective, because this avoids the need for maximum weight restrictions to safeguard spawning animals moving from closed areas during migration, protects both genders from exploitation, and overcomes problems associated with diffusion. However, area closure would have distributional effects through altering the provision of fishing opportunities across regions and user groups. These may be large if closed waterways are used. While economic and social costs accruing to the establishment of these area closures are likely to be significant, these closures are of primary importance if harvest is to continue indefinitely.

Open areas should be managed to maximise efficiency to obtain the greatest benefit from the remaining fishery. This analysis demonstrates that individuals within a multiple-cohort semelparous population should ideally be exploited more heavily as they approach spawning age because the user cost accruing to their harvest declines. This indicates the importance of a higher minimum weight limit for the efficient management of both shortfin and longfin populations, but the identification of a single optimal weight requires investigation using a multiple species approach. The importance of no maximum weight restriction is also demonstrated. This is of particular benefit because lower maximum weight limits are inefficient and problematic to implement effectively.

While ITQ systems are inappropriate for extensive application, they would play a key role in the effective allocation of property rights among user groups in open areas. Although uncertainty and longfin biology prevent the calculation of TACs using traditional stock assessment methods (Ricker 1975; Quinn and Deriso 1999), their identification is aided within this integrated management strategy as density-dependent growth and recruitment from closed populations could potentially provide a buffer against their miscalculation. This buffer would provide fishery managers with some scope to explore various rates of exploitation and levels of allocation using active adaptive management. This may be difficult to implement because of user group pressure, environmental variation, and the longevity of longfins. While these limitations may be significant, the use of more straightforward methods is problematic due to their reliance on stock assessment data, which is unavailable for longfins and is problematic to formulate, given this specie's biology.

8.4 Limitations

Knowledge of the limitations of this analysis allows the strength of its conclusions to be appreciated and its usefulness in policy formation to be ascertained. These limitations primarily arise from the void that exists between reality and the bioeconomic modelling of ecological systems.

This model assumes that the stock size is known. This is clearly an abstraction since longfin biology and significant regional heterogeneity prevent its estimation. The effects of historical exploitation are also difficult to ascertain because of these factors and errors in data collection, especially those for catch and effort (Jellyman 1993). These problems are difficult to overcome and decrease the applicability of any suitable exploitation rate identified by management models of freshwater eel fisheries.

Poor information also reduces the applicability of any optimal exploitation rate identified by this model because it is difficult to identify model parameters with any certainty. The importance of noting this limitation is demonstrated in the sensitivity of this rate to parameter change (Section 7.4). Of particular concern is the portrayal of density-dependent growth, as the strength of this relationship plays an important role in this model but is based on simulated data.

This model abstracts away from the multiple species nature of many habitats within this fishery. While this permits greater insight into the optimal management of the longfin, it fails to consider the implications of these management strategies for the shortfin.

This model also does not incorporate variability in recruitment. The incorporation of stochastic recruitment is unlikely to greatly affect model output (Ludwig and Walters 1982; Clark 1985; Conrad and Clark 1987), especially given the greater importance of physical growth to net production.

This analysis also abstracts from reality through ignoring recreational and customary fishers. This model therefore fails to include important non-market values and the preference of these harvesters for larger longfins. It is hypothesised that the incorporation of these user groups would increase the pressure placed on populations of larger longfins and thus reinforce the importance of area closure.

This analysis also assumes that gear is perfectly selective. While this allows the investigation of the implications of a price that varies with size/age, this provides an overly optimistic view of the ability of the sole owner to control spawning biomass, especially given the significant uncertainty present within reality. This reinforces the importance of area closures for rebuilding/maintaining spawning biomass.

This analysis has also used a harvest-stock ratio to represent the cost function for this fishery, instead of a traditional production function based in production theory, for example the Cobb-Douglas specification (Zellner et al. 1966). A more accurate relationship is required, but data constraints presently prevent its estimation.

The limitations in this section arise from strong assumptions that simplify the complexity associated with analysing longfin populations. These limitations raise some questions regarding the validity of recommendations, but these will always be present in analyses that simplify real systems (Clark 1985; Grimm 1999). Significant simplification is required in this analysis, both to maintain mathematical tractability and because of poor information, the latter dictating that all "models of eel population dynamics should, in general, be simple" (Hoyle and Jellyman 2002, p894). While the exact quantitative predictions of this model must be treated with caution because of this abstraction and the sensitivity of conclusions to certain assumptions must be considered, this research nonetheless provides valuable insight into the optimal management of the longfin eel fishery.

8.5 Recommendations for further research

The analysis of suitable management strategies for the New Zealand freshwater eel fishery is problematic because of poor information. Further research is therefore required if the management of this fishery is to become increasingly sustainable.

An area of primary importance is the formulation of stock assessment procedures. Although very difficult, this would help to monitor the state of the fishery and therefore guide the adjustment of TACs as populations fluctuate due to environmental variation and in response to management. A key factor would be improving the collection of effort and species composition data, as this would provide information from which trends within the fishery may be more accurately identified.

The optimal characteristics of areas or waterways for closure need to be identified. This is important because the future strength of longfin populations depends on these systems for the provision of spawners. This importance is magnified in that this is the fifth analysis to identify the need for area closure (see Chisnall and Hicks 1993; Jellyman 1993; Jellyman et al. 2000; and Hoyle and Jellyman 2002 as well).

It is important that the existence of density-dependent growth in longfin populations is verified. If it does exist, and many authors suggest that it does (Chisnall and Hayes 1991; Chisnall and Hicks 1993; Jellyman et al. 1995; Jellyman 1997), then its strength needs to be identified, since this analysis demonstrates the key importance of this factor in the rapid recovery of open fisheries and its potential role as a buffer to the miscalculation of TACs.

A multiple species analysis is required for providing further insight into the efficient management of the New Zealand freshwater eel fishery. This is of particular importance as the minimum weight restriction and optimal exploitation rate identified within this thesis are unsuitable for application to habitats where both species are found together. The quality of insight provided by a multiple species analysis would be aided through the construction of a suitable production function for this fishery.

Research is also required into the motivations of cultural and recreational harvesters and their historic catch. The inclusion of this information in management models would provide a more detailed view of the current state of the fishery and the suitability of alternative management structures.

8.6 Conclusions

The identification of a sustainable management strategy for the longfin eel fishery is difficult due to this animal's unique and complex biology, which increases its sensitivity to exploitation and exacerbates the difficulties associated with data collection. Moreover, multiple user groups place pressure on larger females approaching spawning. This thesis demonstrates that semelparity causes a divergence between those policies that sustain robust levels of spawners and those that maximise economic efficiency. This divergence is promoted in reality, since any harvest-based regulatory system, no matter how conservative, or use of lower maximum weight restriction will be unable to protect breeding populations due to uncertainty, longfin biology, and incentives for harvest.

This analysis identifies the need for an integrated management strategy, incorporating area closures, for rebuilding and maintaining spawning biomass, and the use of ITQ management in open fisheries to aid the allocation of fishing rights among users. Efficient management of these open areas requires a higher minimum weight limit than under the current management system, and no maximum weight restriction. The calculation of sustainable harvest levels remains problematic due to poor information; however, active adaptive management may be used to work towards their identification. This approach might be aided by density-dependent growth, which would assist the recovery of populations if sustainable harvest were overestimated. Additionally, spawners from closed populations would help to safeguard against recruitment overfishing during the investigation of sustainable exploitation rates. This integrated policy represents a biologically sound and economically relevant management strategy that has the potential to sustain longfin populations and their harvest indefinitely.

APPENDICES

APPENDIX 1. THE MEAN LENGTH AND WEIGHT OF INDIVIDUALS IN EACH YEAR CLASS

The mean length and weight statistics for each year class in the analysed population are presented in Table A1.1.

Table A1.1. The estimated mean length and weight statistics of each age group involved within the study.

Age (*k*)	**Length (mm)**	**Weight (g)**	**Source**
0	103.4	1.46	Beentjes et al. (1997)
1	116.9	2.72	Midpoint between year classes zero and two
2	130.3	3.97	Chisnall et al. (2002)
3	166.1	8.82	Chisnall et al. (2002)
4	201.9	16.73	Chisnall et al. (2002)
5	237.7	28.58	Chisnall et al. (2002)
6	273.5	45.28	Chisnall et al. (2002)
7	309.3	67.79	Chisnall et al. (2002)
8	345.1	97.09	Chisnall et al. (2002)
9	380.9	134.22	Chisnall et al. (2002)
10	416.7	180.21	Chisnall et al. (2002)
11	452.5	236.14	Chisnall et al. (2002)
12	488.3	303.14	Chisnall et al. (2002)
13	524.1	382.33	Chisnall et al. (2002)
14	559.9	474.85	Chisnall et al. (2002)
15	595.7	581.90	Chisnall et al. (2002)
16	631.5	704.66	Chisnall et al. (2002)
17	667.3	844.37	Chisnall et al. (2002)
18	703.1	1002.25	Chisnall et al. (2002)
19	738.9	1179.56	Chisnall et al. (2002)
20	774.7	1377.58	Chisnall et al. (2002)
21	810.5	1597.60	Chisnall et al. (2002)
22	846.3	1840.93	Chisnall et al. (2002)
23	882.1	2108.90	Chisnall et al. (2002)
24	917.9	2402.85	Chisnall et al. (2002)
25	953.7	2724.13	Chisnall et al. (2002)
26	989.5	3074.13	Chisnall et al. (2002)
27	1025.3	3454.21	Chisnall et al. (2002)
28	1061.1	3865.79	Chisnall et al. (2002)
29	1096.9	4310.28	Chisnall et al. (2002)
30	1132.7	4789.11	Chisnall et al. (2002)
31	1168.5	5303.71	Chisnall et al. (2002)

APPENDIX 2. THE SURVEY SENT TO COMMERCIAL HARVESTERS

A2.1 Introduction

This section provides an outline to the survey of commercial harvesters described in Section 6.3.3. The formatting of the covering letter, consent form, and survey has been modified to reduce their length for inclusion in this Appendix.

A2.2.1 Covering Letter

Graeme Doole

Department of Applied & International Economics

Massey University

Private Bag 11222

Palmerston North

New Zealand

19 April 2002

Dear Sir/Madam

I am presently studying for my Master of Applied Economics degree at Massey University, and for my thesis I am investigating the sustainable management of the New Zealand longfin eel fishery. For this study I am required to gain some idea of the cost structure of those eel fishers who harvest in the Waikato River. In order to learn more about this, I have constructed a short survey containing questions regarding the general opinions of harvesters in regard to the current condition of the eel fishery, details surrounding their harvesting activity, and their personal characteristics.

Your name and address have been supplied by the Ministry of Fisheries, who indicate that you are presently licensed to fish commercially for eels in the Waikato area. I am therefore inviting you to participate in this survey; it will take approximately fifteen minutes. Your involvement will be valuable for it will play an important role in the investigation of the sustainability of the New Zealand eel fishery and therefore the incomes of eel harvesters like yourself. Even if you are no longer a commercial eel fishermen, please simply answer Question 1 on the attached survey form and return it in the envelope provided.

Participation in this survey is completely voluntary. Therefore if you choose to fill it out, this will indicate that you understand the information in this letter and that you are willing to be a part of this study. Apart from having the right not to participate, you also may decline to answer any particular questions, withdraw from this study at any time, or ask any questions about this survey and research at any time during participation.

Any answers you provide will be treated as strictly confidential, and will only be used in an aggregated form. The returned surveys will be kept in a locked filing cabinet in a locked office until returns are closed. After this, the details of each will be entered onto a computer without identifying individual participants, and the surveys will be destroyed in a secure manner. There will then be no way of linking any answers to those of individual respondents. At the completion of the project all data, excluding analysis of the results, will be destroyed.

Thank you for your time, it is most appreciated. If you require any further information regarding this survey or study, please feel free to contact either myself or my supervisor through any of the means listed below.

Yours sincerely

Graeme Doole

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Massey University

Private Bag 11222

Palmerston North

New Zealand

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Palmerston North

New Zealand

Phone: 06-350-5514 (work)

06-350-5660 (fax)

Email: R.R.Alexander@massey.ac.nz

A2.2.2 Consent Form

I have read the covering letter and have had the details of the study explained to me. My questions have been answered to my satisfaction, and I understand that I may ask further questions at any time. I understand I have the right to withdraw from the study at any time and to decline to answer any particular questions.

I agree to provide information to the researcher on the understanding that my name will not be used without my permission. I understand that the information will be used only for this research and publications arising from it.

I agree to participate in this study under the conditions set out in the covering letter.

Signed: ....

Name: ....

Date: ....

☐ If you would like a summary of the findings of this research, please tick here.

Please return this form along with the survey.

A2.2.3 Survey

Introduction

Within our overall analysis we are investigating current management strategies and conditions in the longfin eel fishery. Through bringing together the most important characteristics of this system and studying them together, we are hoping to identify those policies that may help to protect the fishery while also maximising financial returns to its participants. Through the inclusion of such monetary considerations, this analysis will therefore provide an alternative to the biological investigations of the longfin fishery that have failed to include some estimate of the effect of different policies on the economic return to harvesters. This survey plays an important role within the study for it will help to identify the general opinion of harvesters regarding the condition of the fishery and its current/future regulation, alongside some estimation of the different components of costs that they face. Through providing information that is not currently available to policy-makers, this survey and analysis will therefore contribute to improving the management and sustainability of the eel fishery.

Part One: Your general opinions of the current condition of the freshwater eel fishery

Please circle or tick your response, or fill in details where appropriate.

1. Are you presently a commercial eel harvester?

☐ Yes ☐ No

If "No", please finish here and return this survey. Thank you for your participation.

2. For how many years have you fished commercially for eels?

☐ <5 ☐ 5-9 ☐ 10-14 ☐ 15-19 ☐ 20+

3. Around what proportion of your total harvest comes from the Waikato River?

(Please circle your response.)

0% 10% 20% 30% 40% 50% 60% 70% 80% 90% 100%

4. Can you confidently distinguish between shortfin and longfin eels?

☐ Yes ☐ No

If you have answered Yes for Question 4, please go on to Questions 5, 6, and 7. If you have answered No, please move to Question 8.

5. Around what percentage of your annual eel harvest would consist of longfins?

(Please circle your response.)

0% 10% 20% 30% 40% 50% 60% 70% 80% 90% 100%

6. Have you noticed a change in longfin eel populations in recent years based on the number you have caught? (Please circle your best response.)

1	2	3	4	5	6
Populations Strictly Declining	Populations Declining	No change	Populations Increasing	Populations Strictly Increasing	Don't Know

7. If you answered either 1 or 2 in Question 6, what do you believe to be the primary cause of the decline? Otherwise, please go to Question 8.

☐ high levels of commercial eel fishing

☐ high levels of customary eel fishing

☐ high levels of recreational eel fishing

☐ pollution

☐ drainage of habitat

☐ droughts

8. Have you noticed a change in the abundance of both shortfins and longfins based on the number you have caught in recent years? (Please circle your best response.)

1	2	3	4	5	6
Populations Strictly Declining	Populations Declining	No change	Populations Increasing	Populations Strictly Increasing	Don't Know

9. If you answered either 1 or 2 in Question 8, what do you believe to be the primary cause of the decline?

☐ high levels of commercial eel fishing

☐ high levels of recreational eel fishing

☐ high levels of customary eel fishing

☐ pollution

☐ drainage of habitat

☐ droughts

Part Two: Harvesting details

10. In which month do you generally start fishing for eels?

(Please circle your response.)

Jan Feb Mar Apr May Jun Jul Aug Sept Oct Nov Dec

11. In which month do you generally stop fishing for eels?

(Please circle your response.)

Jan Feb Mar Apr May Jun Jul Aug Sept Oct Nov Dec

12. How many hours a week on average do you spend eel fishing within this season?

☐ <5 ☐ 5-9 ☐ 10-19 ☐ 20-29 ☐ 30-39 ☐ 40+

13. How many nets do you set on an average night?

14. What do you estimate to be the total cost of fishing in the Waikato River for an average night, including fuel and access payments, but disregarding the gear listed in the question above?

15. Which of the following items of equipment do you use, what number of each do you utilise, around how old are they, and what do you estimate to be their replacement value?

Item	**Number of Items**	**Average Age (Yrs)**	**Replacement Value ($)**
Fyke nets			
Hinaki traps			
Small boat			
Utility vehicle			
Portable tanks/buckets for storing eels			
Fixed tanks for storing eels			
Other			
NOTE: More boxes for the "Other" category were provided in the actual survey.			

Part Three: Your Personal Characteristics and Opinions of the Management of the Freshwater Eel Fishery

16. To what degree do you support the use of the following regulatory policies?

(Please circle your best response in each case.)

A. *Minimum size limits.*

1	2	3	4	5	6
Strongly Oppose	Oppose	Neutral	Support	Strongly Support	No Opinion

B. *Maximum size limits.*

1	2	3	4	5	6
Strongly Oppose	Oppose	Neutral	Support	Strongly Support	No Opinion

C. *Restrictions on the annual number of net/nights a harvester may set.*

1	2	3	4	5	6
Strongly Oppose	Oppose	Neutral	Support	Strongly Support	No Opinion

D. *Area closures.*

1	2	3	4	5	6
Strongly Oppose	Oppose	Neutral	Support	Strongly Support	No Opinion

E. Quota Management System.

1	2	3	4	5	6
Strongly Oppose	Oppose	Neutral	Support	Strongly Support	No Opinion

17. To which age bracket do you belong?

☐ 15-24 ☐ 25-29 ☐ 30-39 ☐ 40-49 ☐ 50-59 ☐ 60+

18. What is your highest level of educational attainment?

☐ No Formal Qualification

☐ School Certificate

☐ Sixth Form Certificate

☐ University Bursary

☐ Other School Qualification

☐ Vocational or Trade Qualification

☐ Bachelor or Higher Degree

☐ Other Post-School Qualification

19. What is your average annual income earned from eel fishing over the last year?

☐ 0-$10,000 ☐ $10,001-$20,000 ☐ $20,001-$30,000 ☐ $30,001-$40,000

☐ $40,000+

20. Around what proportion of your household's income comes from your eel fishing?

(Please circle your response.)

0% 10% 20% 30% 40% 50% 60% 70% 80% 90% 100%

21. Do you have any additional comments regarding the management of the eel fishery?

NOTE: Additional space was provided in the actual survey.

Thank you for your participation.

APPENDIX 3. SUMMARY AND ANALYSIS OF SURVEY RESULTS

A3.1 Summary and analysis

Question 1. Are you presently a commercial eel harvester?

Yes 65 percent

No 0 percent

Question 2. For how many years have you fished commercially for eels?

Figure A3.1 indicates that the eel fishers surveyed are, in general, very experienced (84.7 percent have ten years or more experience); this gives some confidence in the accuracy of their replies. This result may reflect some response bias from the survey, as it is conjectured that replies are more likely to be received from experienced fishers, who are more interested in the sustainability of the fishery because of their investment in it.

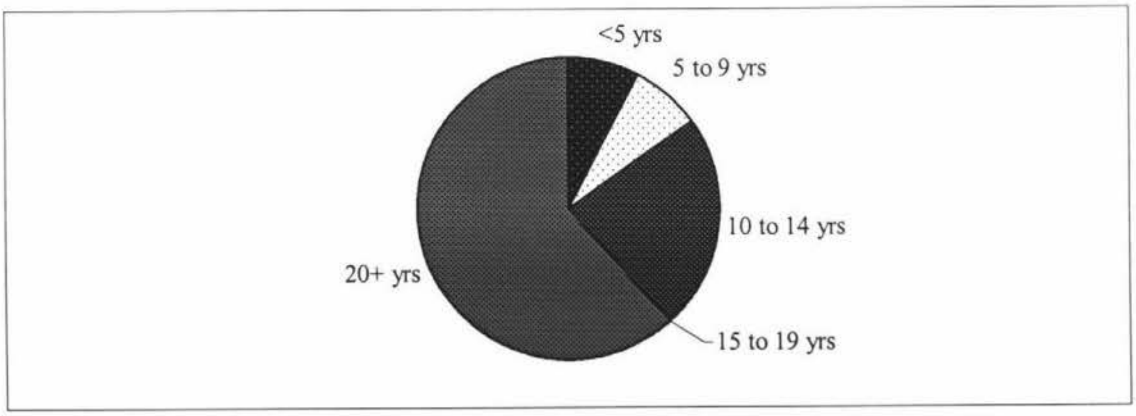

Figure A3.1. Levels of commercial eel fishing experience among harvesters.

Question 3. Around what percentage of your total annual harvest comes from the Waikato River?

The sections of the pie chart presented in Figure A3.2 denote the proportion of harvesters that take a given percentage of annual harvest from the Waikato River.

Figure A3.2 indicates that the majority of harvesters do not take a large proportion of their catch from this waterway. This demonstrates the exploited state of this fishery relative to its condition twenty years ago, as 20 percent of New Zealand's harvest once came from this river (Todd 1981b).

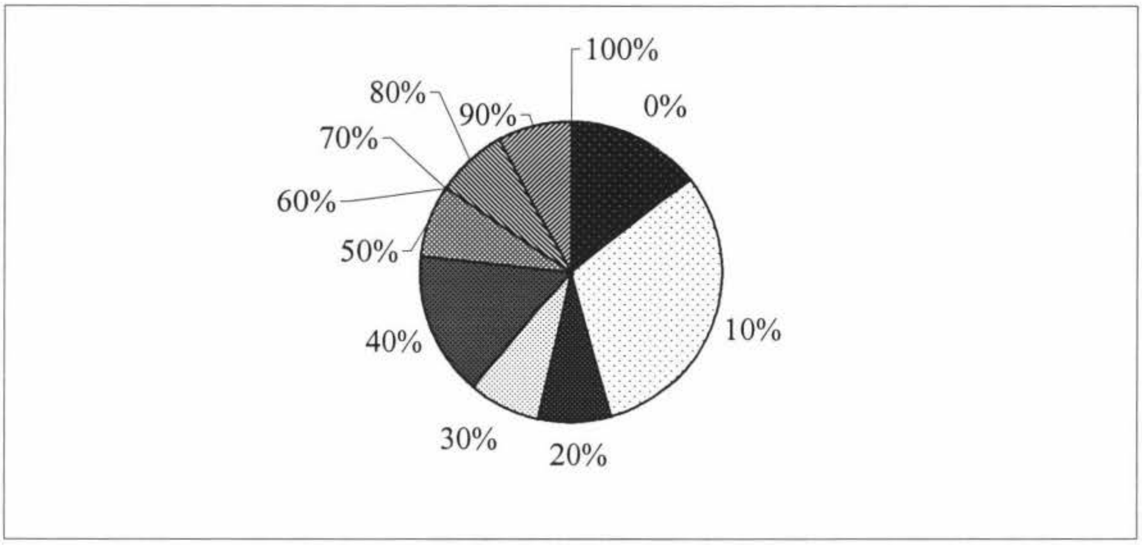

Figure A3.2. The percentage of total annual harvest that fishers take from the Waikato River.

Question 4. Can you confidently distinguish between shortfin and longfin eels?

Yes	65 percent
No	0 percent

This reply is expected, but is necessary before the following questions, which require respondents to make a distinction between these species.

Question 5. Around what percentage of your annual eel harvest would consist of longfins?

The sections within the pie chart presented in Figure A3.3 denote the proportion of harvesters that catch a given level of longfins within their annual harvest. Figure A3.3 indicates a wide distribution across the range of possible answers, with replies ranging from 10 to 90 percent. Although it was initially hypothesised that this pattern arose from the exploited state of the longfin fishery in the Waikato River alone, a linear regression failed to show any significant relationship between the percentage of harvest that a fisher took from the Waikato River and the percentage of longfins within annual catches (R^2=.1788). Respondents catch more shortfins by weight than longfins (Figure A3.3). This demonstrates the strength of shortfin stocks in the Waikato region (Chisnall 1994; Beentjes and Chisnall 1998; Chisnall et al. 2002) in contrast to the historical situation whereby longfins dominated the Waikato region (Chisnall 1994; Chisnall et al. 2002).

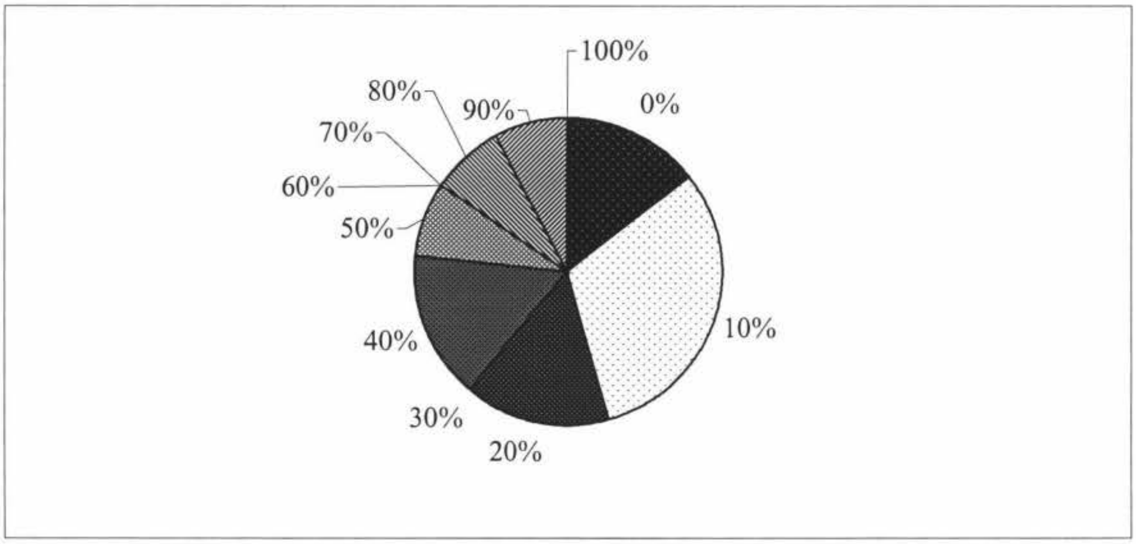

Figure A3.3. The proportion of harvesters that take a certain percentage of longfins each year.

Question 6. Have you noticed a change in longfin eel populations in recent years based on the number you have caught?

Figure A3.4 indicates that there is a significant belief among commercial harvesters that longfin eel populations are declining. Surveyed harvesters may therefore be more willing to accept a possible solution since they are obviously aware of the problem.

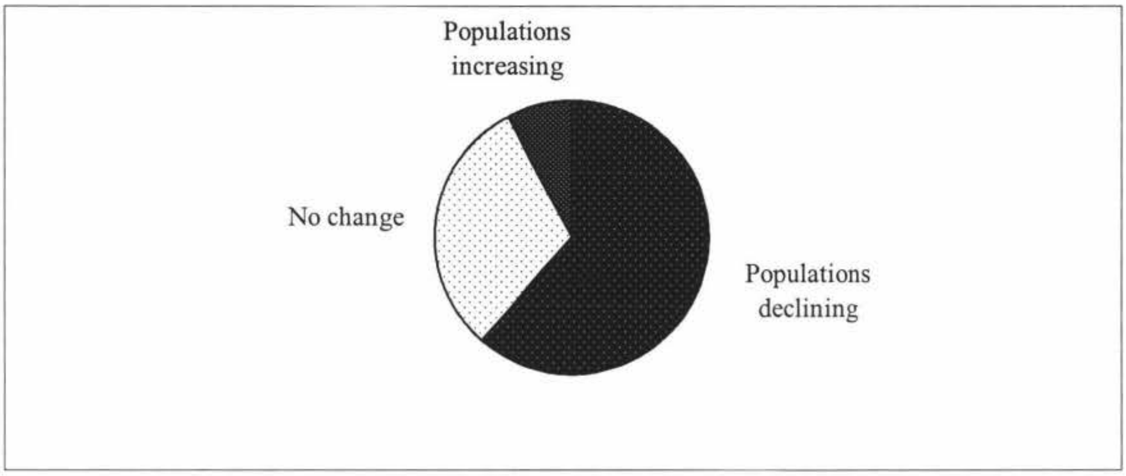

Figure A3.4. The opinions of harvesters regarding changes in longfin eel populations in recent years.

Question 7. If you answered that populations were declining or strictly declining in Question 6, what do you believe to be the primary cause of this?

Although six options were given as possible causes for the decline of longfin populations, only the three included in Figure A3.5 had any response. It is significant that half of those who believe that populations are declining blame commercial eel fishing for this result. This indicates the importance of forming effective property rights in this fishery as a lack of ownership over both eels and access to areas containing them provides no incentive for conservation.

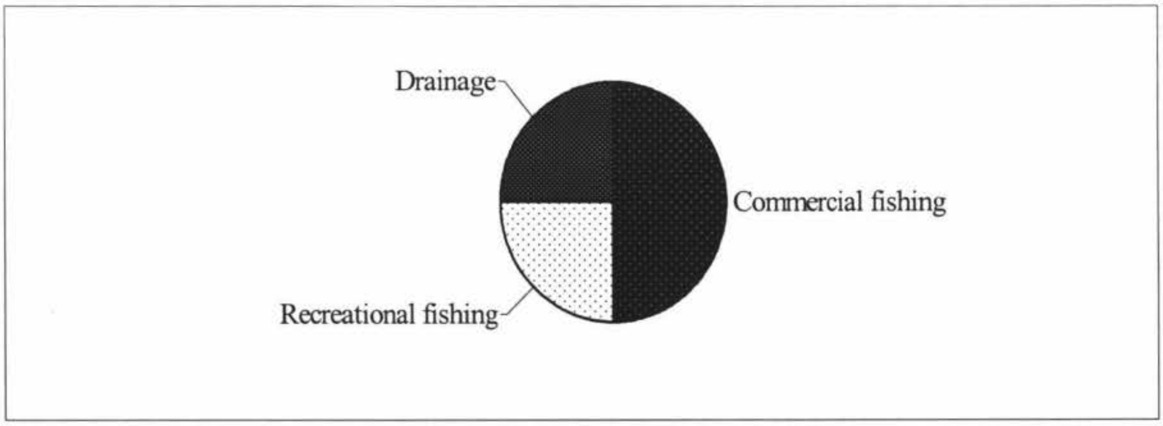

Figure A3.5. The opinions of harvesters regarding the primary cause of decline in longfin eel populations.

Question 8. Have you noticed a change in the abundance of both shortfins and longfins based on the number you have caught in recent years?

Figure A3.6 indicates that a significant proportion of commercial harvesters believe that eel populations are in general decline in the Waikato region, although there is less support for an observable decrease in the entire freshwater eel population than for the longfin alone (61.5 percent versus 46.2 percent) (see Question 6). This indicates the comparative strength of the shortfin fishery.

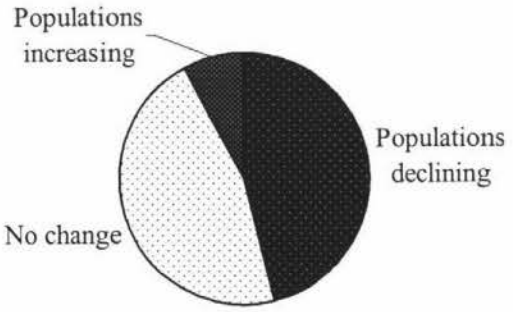

Figure A3.6. The opinions of harvesters regarding changes in both shortfin and longfin populations in recent years.

Question 9. If you answered that populations were declining or strictly declining in Question 8, what do you believe to be the primary cause of this?

Although six options were given as possible causes for the decline of longfin populations, only the three present in Figure A3.7 had any response. There is an even spread between commercial eel fishing, recreational eel fishing, and the drainage of habitat. For more commentary, see Question 7.

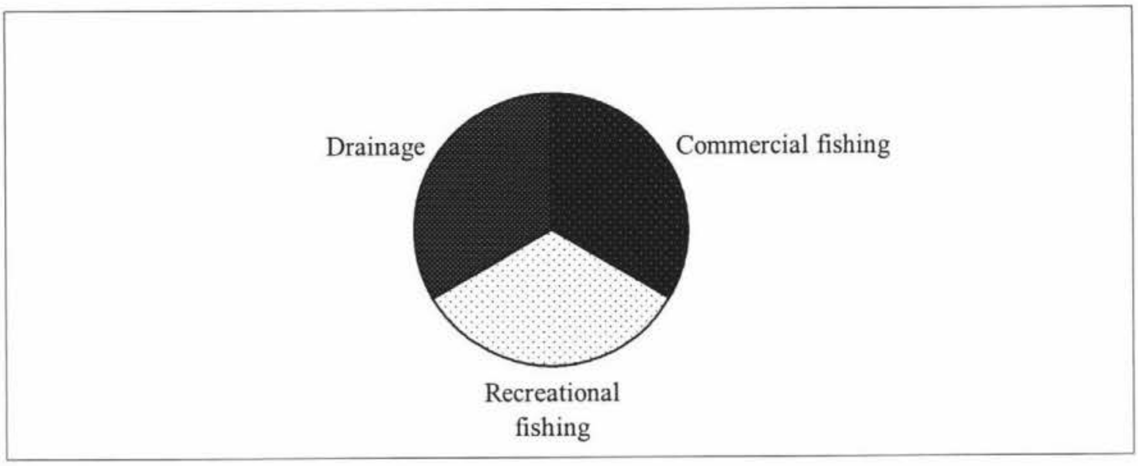

Figure A3.7. The opinions of harvesters regarding the primary cause of the decline in freshwater eel populations.

Question 10. In which month do you generally start fishing for eels?

Question 11. In which month do you generally stop fishing for eels?

While the majority of respondents begin to fish in September and finish in May (61.6 percent), the remainder harvest all year round. This is interesting, because while it is generally believed that it gets too cold or dangerous (due to floods) to harvest eels consistently in winter, a significant proportion of respondents (38.4 percent) do so. This may reflect an inability by these fishers to harvest sufficient weights of eels over the normal season and thus fishing continues throughout the year in a bid to increase returns.

Question 12. How many hours a week on average do you spend eel fishing within this season?

Figure A3.8 indicates that a significant number of surveyed harvesters spend a large part of their working week fishing for eels.

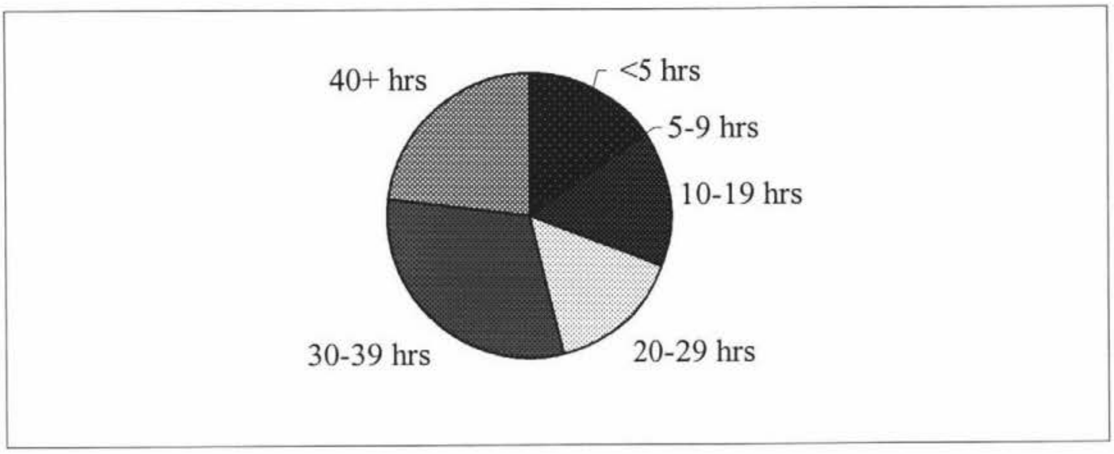

Figure A3.8. Average hours spent eel harvesting within the fishing season.

Question 13. How many nets do you set on an average night?

The number of nets set on average night by the respondents range from 10 nets to 50. These sum to a total of 280 nets, which when divided by the 13 respondents, gives a mean number of nets set on an average night by one individual as 22 nets (rounded due to the discrete nature of the unit).

Question 14. What do you estimate to be the total cost of fishing in the Waikato region for an average night, including fuel and access payments?

The summary for this question and the next outline the calculation of the cost parameter for the model. While this question deals with an estimate of variable costs, Question 15 calculates depreciation and the opportunity cost of harvest.

The primary variable costs of fishers are fuel, bait, and royalty payments to farmers to gain access to waterways containing eels. The variable cost accruing to the harvest of one tonne of eels is calculated using an estimate for the unit cost of effort identified in the survey. This approach provides for increased accuracy, given the ability of respondents to gauge costs more easily for a short period (one night) than for the time required to harvest a tonne, which could take a number of weeks.

The nightly costs of respondents sum to a total of \$505, which when divided by the 13 participants in this survey, gives a mean cost of an average night's harvesting by one individual as \$38.85 (2dp) (standard deviation=\$15.02). Given that the mean number of nets set on an average night by one individual is 22 nets (see Question 13), the variable cost portion of the unit cost of effort is given by \$38.85/(22 nets/night), which is \$1.77/net/night. The mean CPUE for the Waikato region for 1990 to 1999 was 2.9kg/net/night (Ministry of Fisheries CELR data), which corresponds to an average of 345 net nights per harvested tonne. Multiplying \$1.77/net/night by 345 net nights/tonne, it follows that variable cost is \$610.65/tonne.

Question 15. Which of the following items of equipment do you use, what number of each do you utilise, around how old are they, and what do you estimate to be their replacement value?

The items of gear used by the respondents to the survey were utility vehicles, small boats, boat trailers, outboard motors, fyke nets, stakes for holding nets in place, net bags, wetsuits, and eel bags. To identify an annual estimate of depreciation, straight-line depreciation rates are identified for the first four items listed above (see New Zealand Inland Revenue Department 2002). The latter five are not depreciated as these are individually worth less than 200 dollars and thus do not fit the definition of capital items (New Zealand Inland Revenue Department 2002). No annual loss to depreciation is experienced because of the low capital outlay required for harvest (especially in comparison to many ocean fisheries), the significant age of equipment used, and the depreciation rates provided. This is demonstrated, for example, in that the average initial value of utility vehicles is \$30,625, the mean age of these is eleven years, and the straight-line depreciation rate is 18 percent (New Zealand Inland Revenue Department 2002). Accordingly, no depreciation is included within the cost parameter.

The list of capital expenditure provided above demonstrates that the initial outlay to fish is relatively low, this remaining so even when the purchase of nets is considered. For example, each respondent owned an average of forty-eight nets, which at the mean value of \$120 identified within the survey requires an additional investment of only \$5760. These low costs of entry provide some insight into the economic factors (that is, a significantly high price/cost ratio) that promoted the overexploitation of the longfin eel before the exclusion of part-time harvesters in 1984, especially throughout the 1970s when the price for eels was significant (McDowall 1990).

The opportunity cost of harvester time is identified using data on the hourly incomes of New Zealand workers who were paid salaries and wages in the year preceding June 2001 (Table 10, Statistics New Zealand 2002a). This data is drawn from a sample (the New Zealand Income Survey) (Statistics New Zealand 2002b), containing 15,000 private households, administered by Statistics New Zealand. Analysis of how hourly incomes vary indicate that there is little difference between financial return to the age-groups represented in the harvester survey when the five-year age brackets within the Statistics New Zealand survey are amalgamated to form decade-long series like that in the harvester questionnaire. It therefore follows that educational background is used as a proxy for the level of income earned from alternative employment and age is not considered.

First, to find the opportunity cost of harvester time the qualifications held by each respondent were identified from Question 18; these are presented in Column 1 of Table A3.1. An estimate of the number of hours per week that each harvester spends eel fishing within the season is then calculated from Question 12 (Column 2). As this question involved the respondents having to identify an appropriate *range* of hours (for example, 10-19 hours) (see Question 12 above) in order to make the survey more user-friendly, the rounded midpoint of each series (for example, 25 hours for the series 20-29 hours) is used. Responses to the first (<5 hours) bracket were not rounded due to its short length relative to the other ranges, and responses to the last (40+) were ascribed 45 hours, seeing that the midpoint of the three previous ranges involved increases by ten.

Table A3.1. The calculation of the opportunity cost of harvester time.

Qualification	Hours Spent Eel Fishing Per Week	Pre-Tax Mean Hourly Wage ($)	After-Tax Mean Hourly Wage ($)	After-Tax Mean Weekly Wage ($)	No. of weeks in which eel fishing is done per year	Opportunity cost of fishing per haverster ($)
No Qual.	2.5	12.51	10.07	25.18	49	1,233.82
No Qual.	2.5	12.51	10.07	25.18	39	982.02
No Qual.	15	12.51	10.07	151.06	39	5,891.34
No Qual.	15	12.51	10.07	151.06	39	5,891.34
No Qual.	35	12.51	10.07	352.47	39	13,746.33
No Qual.	35	12.51	10.07	352.47	39	13,746.33
No Qual.	35	12.51	10.07	352.47	49	17,271.03
No Qual.	45	12.51	10.07	453.17	49	22,205.33
School Cert.	25	13.63	10.97	274.30	39	10,697.70
School Cert.	45	13.63	10.97	493.75	49	24,193.75
6th Form Cert.	2.5	15.34	12.35	30.87	39	1,203.93
Univ. Bursary	45	13.02	10.48	471.65	39	18,394.35
Univ. Bursary	35	13.02	10.48	366.84	49	17,975.16
KEY: No Qual.=No Qualification Cert.=Certificate Univ.=University					**Total Wages**	153,432.43
					Mean Wage	11,802.49

The mean pre-tax hourly income for each level of educational attainment is then identified from Table 10 of the New Zealand Income Survey (Statistics New Zealand 2002a) described above (Column 3). This is then adjusted for tax using the appropriate tax rates (Taxsaver 2002), to give Column 4. This is then multiplied by the number of hours each respondent spent fishing in a standard week (Column 2) to give the After-Tax Mean Weekly Wage (Column 5). The number of weeks that each harvester spends eel fishing annually is then identified from Questions 10 and 11 (Column 6). It is assumed that fishers who harvest throughout the year have three weeks holiday as specified in legislation (Holidays Act 1981). Column 6 is then multiplied by the After-Tax Mean Weekly Wage (Column 5) to give the opportunity cost of fishing in terms of alternative income for each individual fisher (Column 7). These are summed and the mean is taken to give the mean opportunity cost of each harvester's fishing per year as $11,802.49.

Using unpublished catch data provided by the Ministry of Fisheries, which states that 153.3 tonnes were harvested in the 1999/2000 season in the Waikato region and twenty-three harvesters were licensed to fish for them in this area at that time, it is identified that each licensed fisher had a mean annual harvest of 6.67 tonnes (2dp). Although a mean from a series of years would have been preferable, this fishing year (1999/2000) is the only for which corresponding levels of harvest and numbers of licensed fishers could be obtained. Dividing the mean opportunity cost of each harvester's fishing year from above ($11,802.49) by this amount (6.67 tonnes) gives an approximation of foregone income per harvested tonne of $1769.49 (2dp). Added to the variable cost component provided above ($610.63/tonne), this gives a total harvesting cost of $2380.12 per tonne.

Question 16. To what degree do you support the use of the following regulatory policies?

A. Minimum size limits.

Figure A3.9 indicates full support for minimum size restrictions among survey participants.

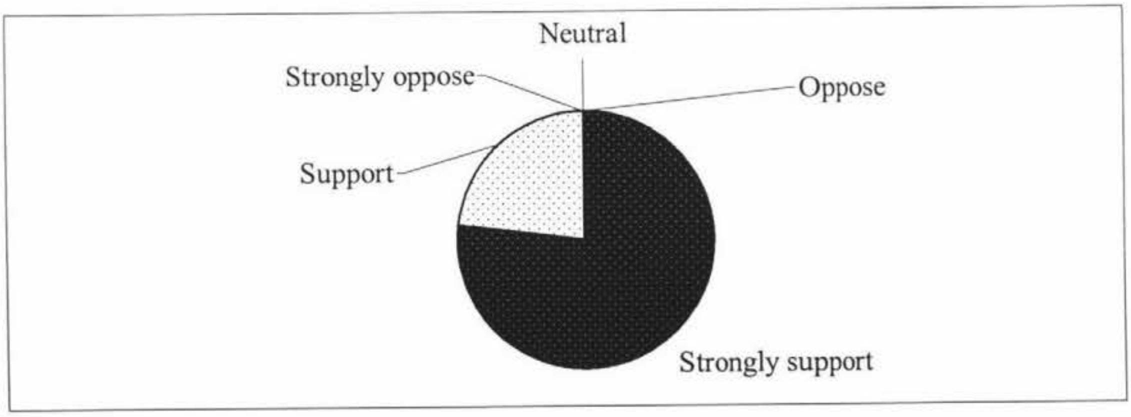

Figure A3.9. Degree of support for the use of minimum size restrictions.

B. Maximum size limits.

In contrast to the use of minimum size restrictions presented above, there is significant opposition to the use of maximum size limits (61.6 percent either oppose or strongly oppose) (Figure A3.10). This reflects the effect this restriction would have on the profit of harvesters, as it will leave less animals in the higher-price bracket available to the fishery.

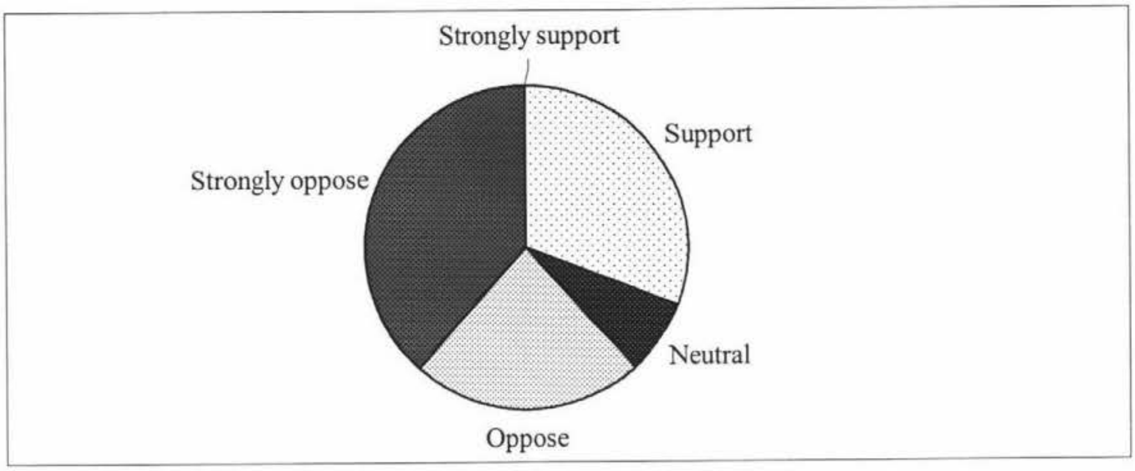

Figure A3.10. Degree of support for the use of maximum size limits.

C. Restrictions on the annual number of net/nights a harvester may use.

Figure A3.11 demonstrates that there is significant opposition to the use of effort restrictions to protect the longfin fishery. These would also be difficult to enforce. Effort restrictions are therefore not considered as a viable future management option within this analysis.

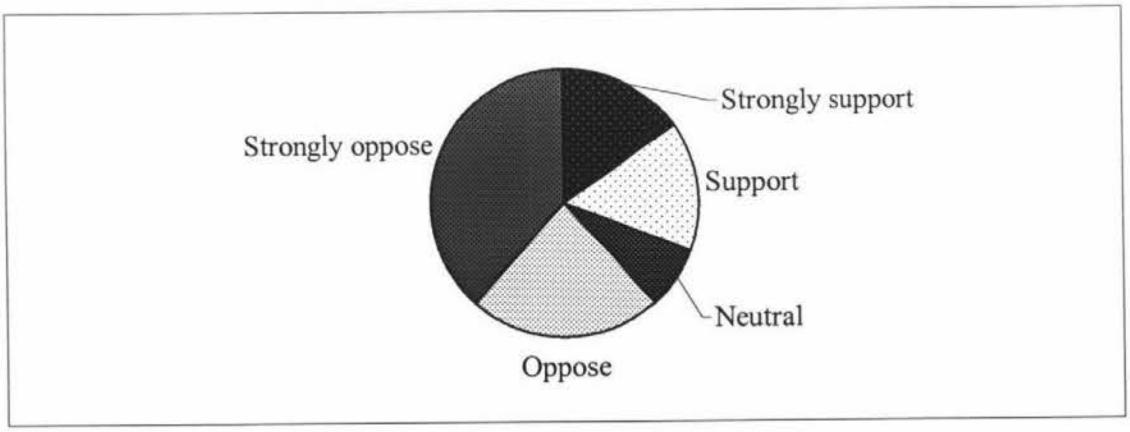

Figure A3.11. Degree of support for the use of effort restrictions.

D. Area closures.

There is low support for area closures among respondents (Figure A3.12), as expected. This indicates at a small scale that such closures would intuitively have negative support from the commercial eel industry.

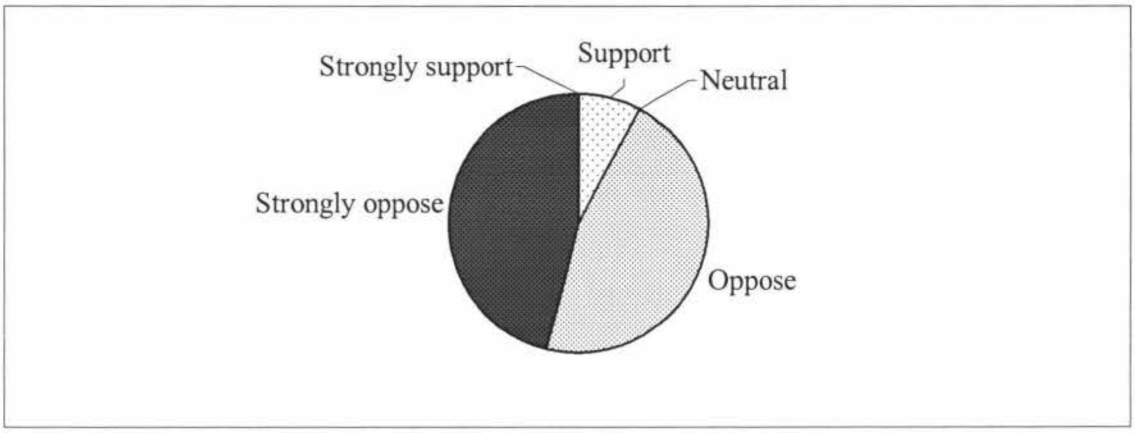

Figure A3.12. Degree of support for the use of area closures.

E. Quota Management System.

Figure A3.13 indicates that there is strong support for the use of the QMS to manage the eel fishery. This is obviously beneficial for the introduction of quotas in the North Island, although the potential limitations of this form of management for the freshwater eel fishery (see Section 3.6.5) may not be widely appreciated.

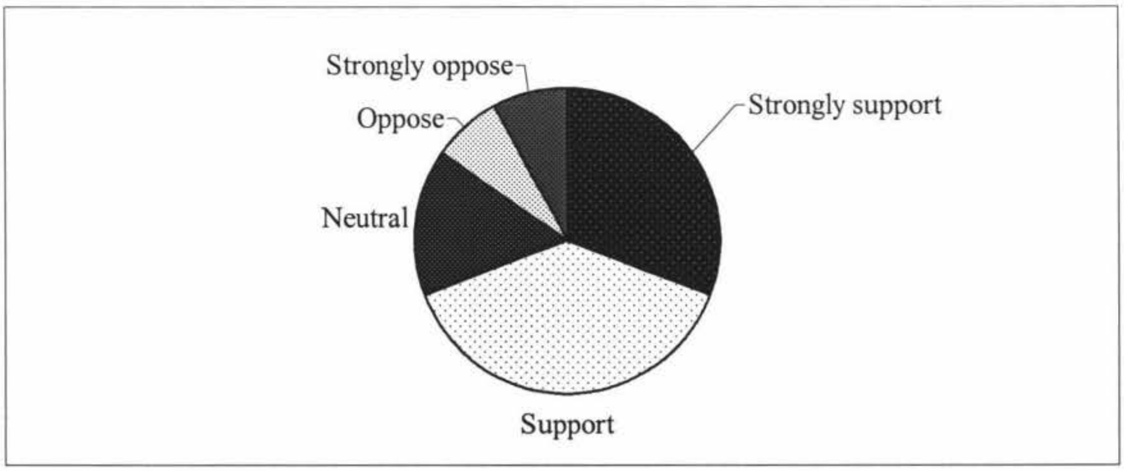

Figure A3.13. Degree of support for the use of ITQs.

Question 17. To which age bracket do you belong?

Figure A3.14 indicates that all the respondents to the survey were either thirty years of age or above, with 46.2 percent being above the age of fifty.

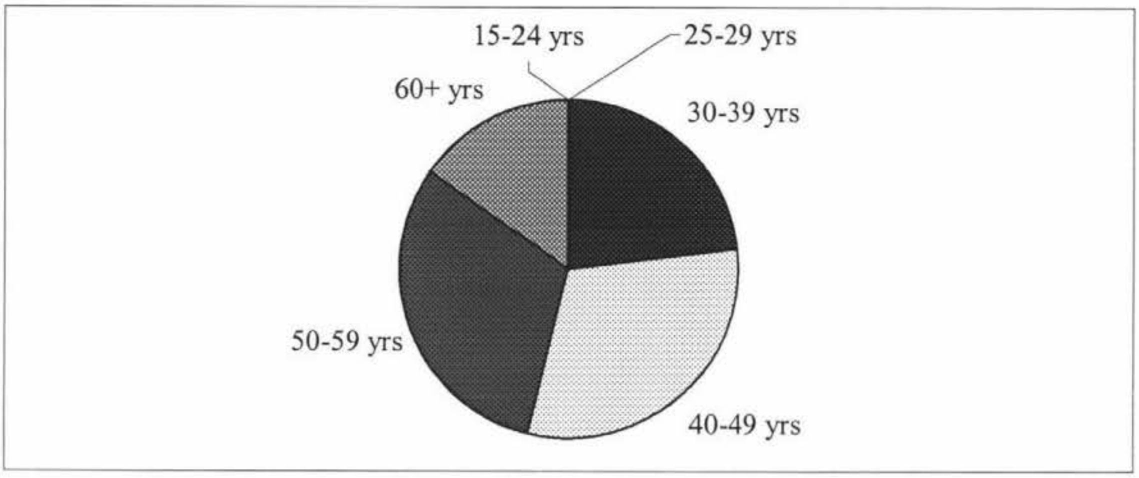

Figure A3.14. The percentage of respondents within each age bracket.

Question 18. What is your highest level of educational attainment?

Figure A3.15 indicates that over half of those who replied to the survey have no formal qualification. This suggests that the alternative employment opportunities available to harvesters are more likely to involve low-wage vocations (Statistics New Zealand 2002a).

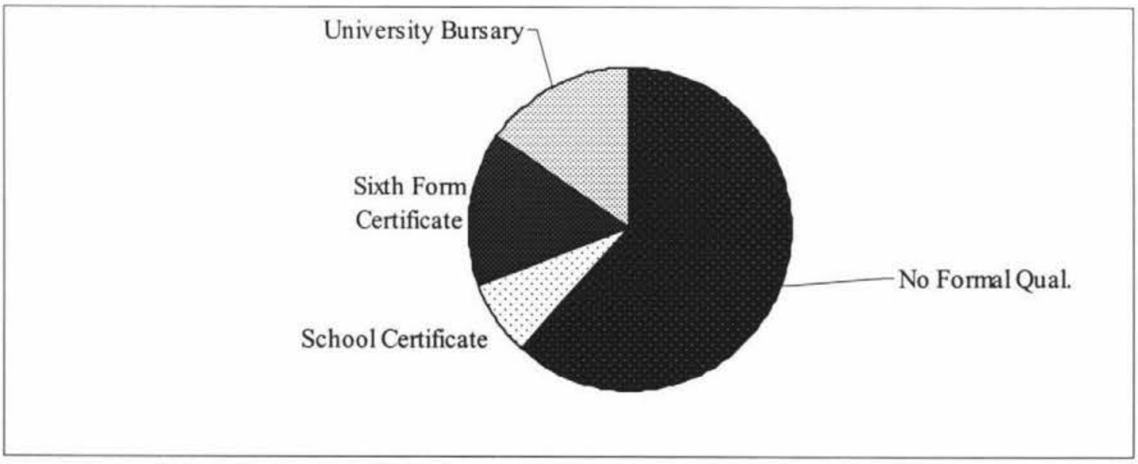

Figure A3.15. The educational attainment of respondents.

Question 19. What is your average annual income earned from eel fishing over the last year?

Figure A3.16 indicates that the majority of respondents (69.3 percent) have earned $20,000 or less from eel fishing over the last year. This reflects that alternative employment opportunities are more likely to involve low-wage vocations, consistent with the findings in Question 18 above, because if higher wages were available elsewhere, then economic theory postulates that harvesters would seek that alternative employment.

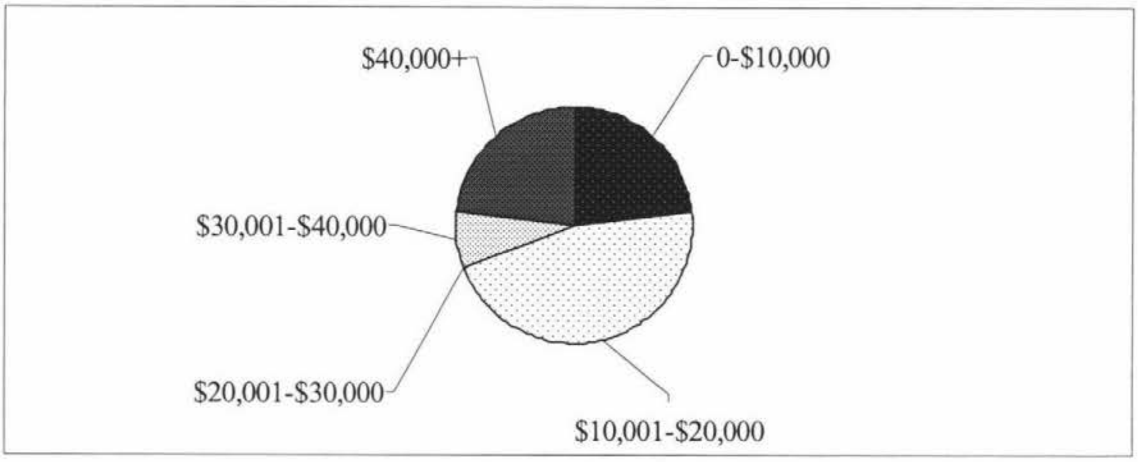

Figure A3.16. Income earned by respondents from eel fishing over the last year.

Question 20. On average, around what percentage of your household's income comes from your eel fishing?

The sections within the pie chart presented in Figure A3.17 denote the proportion of harvesters that earn a certain percentage of household income from eel fishing per year. Figure A3.17 indicates that while a significant fraction is earned from eel harvesting in many cases, the households of survey participants obtain the majority of their income from alternative sources. This figure and information on alternative employment opportunities (Question 18) suggest that if appropriate management strategies are not introduced and longfin populations are permitted to decline further, then a number of households will be adversely affected. Yet a generally low reliance on eel fishing for a significant proportion of household income (Figure A3.17) suggests that the social costs of limiting harvest to permit recovery could be lower than expected, especially since this would, it is hoped, build a fishery of greater future worth.

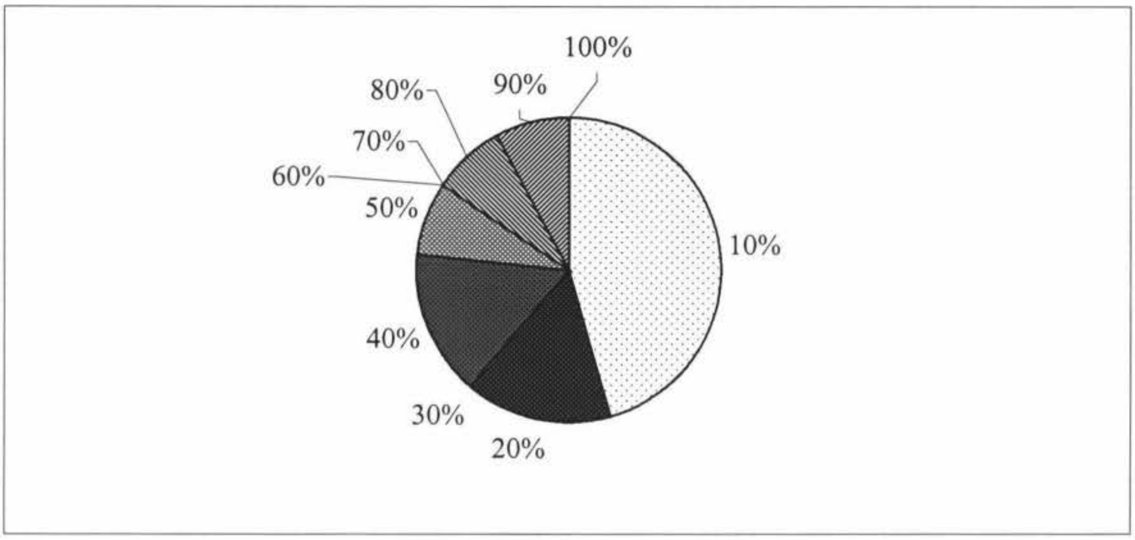

Figure A3.17. The mean percentage of annual household income earned from eel fishing by respondents.

Question 21. Do you have any additional comments regarding the management of the eel fishery?

A number of interesting comments regarding the management of the eel fishery were made. Respondents often commented on the lack of incentives for conservation offered within the current management structure and the subsequent urgency with which the quota system should be introduced. The voluntary use of a higher minimum weight in waterways where fishers had been granted sole access was also mentioned.

A3.2 Conclusions

In conclusion, it seems that commercial harvesters are aware of the decline observed in longfin populations by biologists (Question 6) and have an understanding that commercial fishing has played a role (Question 7). There is strong support for both minimum weight restrictions and the QMS, indicating that many fishers are confident that the enhancement of the eel fishery requires a reassessment of current management. These findings reinforce the need for the analysis of suitable management strategies in this thesis.

APPENDIX 4. CONCISE SPECIFICATION OF THE GAMS PROGRAMME

```
$title baseline eel model
$offsymxref offsymlist
options solprint=off, limrow=0, limcol=0, decimals=1, reslim=5000;
set  t time periods/a-1*a-13,0*65/
    k age groups/11*31/
    o price increasing in age and losses to mortality and migration /pr,mo/;
parameter df(t) discount factor;
df(t)=(1.0/(1.0+.05))**(ord(t)-1)$(ord(t) gt 13);
table mp(k,o)  price and nat mortality
         pr      mo
11       0.435  -0.04
12       0.435  -0.055
13       0.435  -0.075
14       0.435  -0.145
15       0.454  -0.154
16       0.454  -0.217
17       0.454  -0.097
18       0.485  -0.065
19       0.485  -0.04
20       0.485  -0.04
21       0.595  -0.04
22       0.595  -0.04
23       0.595  -0.04
24       0.595  -0.04
25       0.595  -0.04
26       0.595  -0.04
27       0.595  -0.04
28       0.595  -0.04
29       0.595  -0.04
30       0.595  -0.04;
scalar ylim harvest limit/1/;
scalar x0/6.9/;
scalar xsp/.00599/;
scalar c cost per unit effort/.238012/;
scalar i stock externality exponent/1/;
scalar a/1.4/;
scalar b/6.7/;
scalar gm/4.65/;
scalar ddg/.04/;
alias (k,age);
positive variables
x(k,t) stock in cohort k in time t
y(k,t) harvest of cohort k in time t
xs(t) total stock in time t
ys(t) total harvest in time t;
```

```
free variable
sw aggregate social welfare;
equations      sweq objective function
               recruit(k,t) recruitment
               growth(k,t) resource growth
               totalst(t) total stock
               envcon(t) environmental constraint
               totalharv(t) total harvest
               harvcon(k,t) harvest constraint
               quotacon(t) quota constraint;
sweq.. sw =e= sum(t$(ord(t) gt 13),df(t)$(ord(t) gt 13)*sum(k$(ord(k) lt
21),y(k,t)*(mp(k,"pr")-c*(y(k,t)/(x(k,t)**i)))));
recruit("11",t+13)..     x("11",t+13) =e= (a*x("31",t))/(1+b*x("31",t))*exp(gm)*1/100;
growth(k+1,t+14)$(ord(k) lt 21)..     x(k+1,t+14) =e= (x(k,t+13)*exp(ddg*(x0-
sum(age,x(k,t+13)))+mp(k,"mo"))-y(k,t+13));
totalst(t)..             xs(t) =e= sum(k,x(k,t));
envcon(t)..              x0 =g= xs(t);
totalharv(t)..           ys(t) =e= sum(k$(ord(k) lt 21),y(k,t));
harvcon(k,t)..           x(k,t) =g= y(k,t);
quotacon(t)..            ylim*xs(t) =g= ys(t);
x.lo(k,t) = 0.002;
x.fx("31",t)$(ord(t) lt 14) = .00599;
x.fx("11","0") = ((a*xsp)/(1+b*xsp))*exp(gm)*1/100;
x.fx("12","0") = 0.0849;
x.fx("13","0") = 0.0795;
x.fx("14","0") = 0.0683;
x.fx("15","0") = 0.0574;
x.fx("16","0") = 0.0448;
x.fx("17","0") = 0.039;
x.fx("18","0") = 0.0346;
x.fx("19","0") = 0.0313;
x.fx("20","0") = 0.0281;
x.fx("21","0") = 0.0251;
x.fx("22","0") = 0.0222;
x.fx("23","0") = 0.0196;
x.fx("24","0") = 0.0171;
x.fx("25","0") = 0.0149;
x.fx("26","0") = 0.0129;
x.fx("27","0") = 0.0112;
x.fx("28","0") = 0.00962;
x.fx("29","0") = 0.00824;
x.fx("30","0") = 0.00704;
x.fx("31","0")= 0.00599;
model eel1/all/;
eel1.optfile=1;
set lfnsup := 5000;
option nlp = conopt2;
option iterlim = 25000;
solve eel1 using nlp maximising sw;
parameter opt1 optimal total values;
```

```
parameter opt2 optimal cohort values by k and t;
opt1(t,"opt.totst")=xs.l(t)$(ord(t) gt 13)*100;
opt1(t,"opt.totha")=ys.l(t)$(ord(t) gt 13)*100;
opt1(t,"sp.biom.")=x.l("31",t)$(ord(t) gt 13)*100;
opt2(k,t,"opt.stock")=x.l(k,t)$(ord(t) gt 13)*100;
opt2(k,t,"opt.harv")=y.l(k,t)$(ord(t) gt 13)*100;
display opt1, opt2;
```

APPENDIX 5. DETAILED DESCRIPTION OF THE GAMS PROGRAMME

A4.1 Introduction

In this appendix a detailed description of the programme used to formulate the base run of the model in GAMS is provided. Commentary is provided in italicised text. Section 6.8 provides a summary of all exogenous data and the numerical model used. For a detailed description of GAMS commands, see Brooke et al. (1988).

A4.2 Baseline model

These commands limit diagnostic output after model solution has been secured in previous runs.

```
$offsymxref offsymlist
```

The first three commands that follow the options *command also limit diagnostic output, while the* decimals *command limits the decimal places reported in the solution file.*

```
options solprint=off, limrow=0, limcol=0, decimals=1;
```

Now the indices (sets) for the problem are defined. The first thirteen time periods are labelled differently as they are merely initial periods to allow for the solution of the model and do not contribute to the study period (see Section 6.8). The third set (o) has two separate elements (pr and mo) that allow the formation of a table describing features that vary by age (k) (price and natural mortality/male migration).

```
sets t time periods/a-1*a-13,0*65/
     k age groups/11*31/
     o price increasing in age and losses to mortality and migration/pr,mo/;
```

The parameter command below defines the discount factor, where r=.05.

```
parameter df(t) discount factor;
df(t)=(1.0/(1.0+.05))**(ord(t)-1)$(ord(t) gt 13);
```

The table below defines the price values and mortality/migration figures for each age cohort. Given the sensitivity of GAMS's NLP solvers to correct scaling, stock and harvest variables in the model are expressed in units of one hundred tonnes, while the objective function is presented in millions of dollars. The price data is therefore multiplied by one hundred and then divided by one million to get the values below.

```
table mp(k,o) price, natural mortality and male migration
        pr     mo
11      0.435  -0.04
12      0.435  -0.055
```

```
13      0.435  -0.075
14      0.435  -0.145
15      0.454  -0.154
16      0.454  -0.217
17      0.454  -0.097
18      0.485  -0.065
19      0.485  -0.04
20      0.485  -0.04
21      0.595  -0.04
22      0.595  -0.04
23      0.595  -0.04
24      0.595  -0.04
25      0.595  -0.04
26      0.595  -0.04
27      0.595  -0.04
28      0.595  -0.04
29      0.595  -0.04
30      0.595  -0.04;
```

The following define the virgin biomass and the limits to biomass and harvest that will enter into the feasibility constraints. Ylim specifies the highest exploitation rate permitted (as long as $X_{k,t} \geq Y_{k,t}$*) and will change with different quota levels.*

```
scalar ylim harvest limit/1/;
scalar x0 virgin biomass/6.9/;
```

The next scalar provides the initial spawning stock at a 20 percent rate of exploitation (see Section 6.8).

```
scalar xsp initial spawning stock/.00599/;
```

Cost per tonne of harvested eels is multiplied by one hundred and then divided by one million, as described above for price.

```
scalar c cost of harvest/.238012/;
```

The strength of diminishing returns to the population level for stock-related costs is set to unity for the base model.

```
scalar i/1/;
```

To allow for the correct specification of the spawner-recruitment function, the Beverton-Holt parameters are multiplied by one hundred as all stock and harvest variables are measured in hundreds of tonnes.

```
scalar a/1.4/;
scalar b/6.7/;
```

The following command defines the exponent for juvenile survival.

```
scalar gm/4.65/;
```

The next scalar defines the exponent for density-dependent growth. This is also multiplied by one hundred in order to account for scaling.

```
scalar ddg/.04/;
```

The following command merely assigns another name for the set k. This is needed to allow for the definition of density-dependent growth.

```
alias (k,age);
```

The following defines the strictly positive variables.

```
positive variables
x(k,t) weight of stock in cohort k in time t
y(k,t) harvest of cohort k in time t
xs(t) total stock in time t
ys(t) total harvest in time t;
```

The following ensures that the objective function can be both positive and negative.

```
free variable
sw definition of social welfare;
```

The following declares the equations presented in the model.

```
equations
              sweq objective function
              recruit(k,t) recruitment constraint
              growth(k,t) growth constraint
              totalst(t) definition of total stock
              envcon(t) environmental constraint
              totalharv(t) definition of total harvest
              quotacon(t) quota constraint
              harvcon(k,t) harvest feasibility constraint;
```

The following defines the equations of the model, where =E= means strictly equal to and =G= means greater than or equal to. The equation SWEQ is the objective function presented in equation (6.45). The equation RECRUIT is the recruitment constraint presented in equation (6.46). The equation GROWTH is the growth constraint presented in equation (6.47). The term $(ORD(K) lt 21) within this equation specifies that the growth constraint is only valid for those cohorts up to and including k=30, given that the last age class is k=31. The equation TOTALST defines the total adult stock, defined in equation (6.54). The equation ENVCON defines the feasibility constraint that defines that the total level of stock can never be greater than virgin biomass, as outlined in equation (6.50). The equation TOTALHARV defines the total level of harvest, according to equation (6.55). The equation QUOTACON defines the quota constraint outlined in equation (6.49). The YLIM scalar identified above is set to one for model runs incorporating unconstrained harvest. The equation HARVCON *defines the feasibility constraint stating that harvest must be less than or equal to the stock in all periods. This is outlined in equation (6.53). The definitions of each equation are:*

```
sweq.. sw =e= sum(t$(ord(t) gt 13),df(t)$(ord(t) gt 13)*sum(k$(ord(k) lt
21),y(k,t)*(mp(k,"pr")-c*(y(k,t)/(x(k,t)**i)))));
recruit("11",t+13)..  x("11",t+13) =e= (a*x("31",t))/(1+b*x("31",t))*exp(gm)*1/100;
growth(k+1,t+14)$(ord(k) lt 21)..    x(k+1,t+14) =e= (x(k,t+13)*exp(ddg*(x0-
sum(age,x(k,t+13)))+mp(k,"mo"))-y(k,t+13));
totalst(t)..              xs(t) =e= sum(k,x(k,t));
envcon(t)..               x0 =g= xs(t);
totalharv(t)..            ys(t) =e= sum(k$(ord(k) lt 21),y(k,t));
quotacon(t)..             ylim*xs(t) =g= ys(t);
harvcon(k,t)..            x(k,t) =g= y(k,t);
```

Although not present in this base model, to restrict the harvest of any cohorts to zero to represent alternative weight restrictions (equation (6.48)) it is necessary to specify two equations under the equations *command above, one for a minimum legal weight and one for a maximum legal weight. For example:*

```
agemin(k,t) minimum weight constraint
agemax(k,t) maximum weight constraint
```

The following equations are then added in the block of defined equations:

```
agemin(k,t)$(ord(k) lt 5)..    Y(K,T) =E= 0;
agemax(k,t)$(ord(k) gt 12)..  Y(K,T) =E= 0;
```

The ord(k) commands for each minimum weight restriction are presented in Table A5.1. These specify that the constraint must hold for all cohorts beneath these year classes. For example, ord(k) lt 5 *means that harvest must be zero for all cohorts beneath the fifth in the series k=[11,...,31], which are k=[11,12,13,14]. The ages corresponding to each limit are provided in Table 6.6.*

Table A5.1. Ord(k) commands for each minimum weight restriction.

Minimum weight restriction (kg)	**GAMS command**
.5	ord(k) lt 5
1	ord(k) lt 8
1.5	ord(k) lt 11

The ord(k) commands for each maximum weight limit are presented in Table A5.2. In contrast to those for the minimum weight restrictions, these specify that the constraint must hold for cohorts above the specified cohort. For example, ord(k) gt 18 means that harvest must be zero for all cohorts above the eighteenth in the series k=[11,...,31], which are k=[29,30,31].

Table A5.2. Ord(k) commands for each maximum weight restriction.

Maximum weight restriction (kg)	**GAMS command**
2	ord(K) gt 12
3	ord(K) gt 15
4	ord(K) gt 18
-	-

The following command sets a lower bound on cohort stocks, while the constraint "envcon(t)" bounds the stock level through prohibiting the total stock to go above the virgin biomass. The minimum stock level is set at .2 tonnes, as a zero value prevents the GAMS solver from completing normally. This occurs as all relevant partial derivatives consequently become zero and, since harvest must be less than or equal to the stock in each period, it will be immediately optimal to harvest the only feasible yield (zero tonnes) defined by this condition. This causes the objective function to equal zero without any search for alternative solutions taking place. Setting the stock to this minimum level therefore overcomes this problem, without any loss of realism. The inclusion of the constraint prohibiting the total stock to go above virgin biomass, the density-dependent spawner-recruitment and growth relationships, and a positive stock-related cost externality also aid the algorithm by preventing it from moving towards regions where derivatives are large.

```
x.lo(k,t) = 0.002;
```

The following provides the initial conditions for the model. It is assumed within the model that the population is in equilibrium at an exploitation rate of 20 percent (see Section 6.8).

```
x.fx("31",t)$(ord(t) lt 14) = .00599;
x.fx("11","0") = ((a*xsp)/(1+b*xsp))*exp(gm)*1/100;
x.fx("12","0") = 0.0849;
x.fx("13","0") = 0.0795;
x.fx("14","0") = 0.0683;
x.fx("15","0") = 0.0574;
x.fx("16","0") = 0.0448;
x.fx("17","0") = 0.039;
x.fx("18","0") = 0.0346;
x.fx("19","0") = 0.0313;
x.fx("20","0") = 0.0281;
x.fx("21","0") = 0.0251;
x.fx("22","0") = 0.0222;
x.fx("23","0") = 0.0196;
x.fx("24","0") = 0.0171;
x.fx("25","0") = 0.0149;
x.fx("26","0") = 0.0129;
```

```
x.fx("27","0") = 0.0112;
x.fx("28","0") = 0.00962;
x.fx("29","0") = 0.00824;
x.fx("30","0") = 0.00704;
x.fx("31","0")= 0.00599;
```

This defines the model "eel" *as all of those equations listed above under the command* equations.

```
model eel/all/;
```

The following helps to overcome any problems associated with slow convergence.

```
eel.optfile=1;
set lfnsup := 2000;
```

The model is solved using Non-Linear Programming (NLP) utilising the CONOPT2 solver, with an iteration limit of 25,000.

```
option nlp = conopt2;
option iterlim = 25000;
solve eel using nlp maximising sw;
```

Values of interest are computed by multiplying model output by one hundred to obtain solution variables in tonnes. These values are then presented using the display *command.*

```
parameter opt1 optimal total values;
parameter opt2 optimal cohort values by k and t;
opt1(t,"opt.totst")=xs.l(t)$(ord(t) gt 13)*100;
opt1(t,"opt.totha")=ys.l(t)$(ord(t) gt 13)*100;
opt1(t,"sp.biom.")=x.l("31",t)$(ord(t) gt 13)*100;
opt2(k,t,"opt.stock")=x.l(k,t)$(ord(t) gt 13)*100;
opt2(k,t,"opt.harv")=y.l(k,t)$(ord(t) gt 13)*100;
display opt1, opt2;
```

This completes the specification of the base run of the model.

APPENDIX 6. NET PRESENT VALUES ACCRUING TO ALTERNATIVE MANAGEMENT POLICIES

The Net Present Values (NPVs) accruing to the alternative management strategies are summarised in Table A6.1. The weight limits in the first column represent a range between a given minimum weight (the first number) and a maximum weight (the second number). "No max." denotes that this policy incorporates no maximum weight restriction.

Table A6.1. The Net Present Values accruing to the analysed strategies.

Weight limits (kg)	**NPV at TAC=5% ($)**	**NPV at TAC=10% ($)**	**NPV with no TAC ($)**
.5 to 2	383,600	476,400	487,100
.5 to 3	470,700	622,100	684,500
.5 to 4	523,200	742,400	796,600
.5, no max.	540,600	810,500	877,600
1 to 2	377,100	461,600	474,100
1 to 3	468,300	614,800	648,500
1 to 4	514,900	737,800	801,700
1, no max.	537,800	807,000	876,000
1.5 to 2	370,500	442,100	457,500
1.5 to 3	468,000	611,500	644,400
1.5 to 4	514,800	737,100	799,900
1.5, no max.	537,700	806,500	874,700

APPENDIX 7. LEVELS OF HARVEST ACCRUING TO ALTERNATIVE MANAGEMENT POLICIES

The equilibrium harvest levels accruing to the alternative management strategies are summarised in Table A7.1. The weight limits in the first column represent a range between a given minimum weight (the first number) and a maximum weight (the second number). "No max." denotes that this policy incorporates no maximum weight restriction.

Table A7.1. Optimal equilibrium harvest levels for the analysed strategies.

Weight limits (kg)	**Harvest at TAC=5% (tonnes)**	**Harvest at TAC=10% (tonnes)**	**Unconstrained harvest**	
			Tonnes	**Rate of harvest (%)**
.5 to 2	16	19	19	11.2
.5 to 3	17	24	22	12.6
.5 to 4	18	24	25	13.9
.5, no max.	18	25	26	14.1
1 to 2	16	20	20	11.6
1 to 3	17	23	23	13.7
1 to 4	18	26	26	15
1, no max.	18	28	28	15.2
1.5 to 2	16	20	20	10.5
1.5 to 3	17	23	22	12
1.5 to 4	18	26	25	13.5
1.5, no max.	18	28	26	14.1

APPENDIX 8. LEVELS OF SPAWNING BIOMASS ACCRUING TO ANALYSED POLICIES

The levels of spawning biomass accruing to the alternative management strategies are summarised in Table A8.1. The weight limits in the first column represent a range between a given minimum weight (the first number) and a maximum weight (the second number). "No max." denotes that this policy incorporates no maximum weight restriction.

Table A8.1. Optimal equilibrium spawning biomass levels for each of the analysed strategies.

Weight limits (kg)	Spawning biomass with TAC=5% (tonnes)	Spawning biomass with TAC=10% (tonnes)	Spawning biomass with no TAC (tonnes)
.5 to 2	22	12	7
.5 to 3	20	12	3
.5 to 4	18	9	2
.5, no max.	15	8	3
1 to 2	21	10	7
1 to 3	20	10	3
1 to 4	20	8	2
1, no max.	15	6	2
1.5 to 2	20	8	8
1.5 to 3	20	10	6
1.5 to 4	18	8	4
1.5, no max.	15	6	3

APPENDIX 9. LEVELS OF STOCK ACCRUING TO ALTERNATIVE MANAGEMENT POLICIES

The population levels accruing to the alternative management strategies are summarised in Table A9.1. The weight limits in the first column represent a range between a given minimum weight (the first number) and a maximum weight (the second number). "No max." denotes that this policy incorporates no maximum weight restriction.

Table A9.1. Levels of equilibrium stock for each of the analysed strategies.

Weight limits (kg)	**Stock level at TAC=5% (tonnes)**	**Stock level at TAC=10% (tonnes)**	**Stock level with no TAC (tonnes)**
.5 to 2	322	193	171
.5 to 3	340	236	174
.5 to 4	356	236	180
.5, no max.	365	254	184
1 to 2	321	195	172
1 to 3	340	235	179
1 to 4	353	261	182
1, no max.	368	278	184
1.5 to 2	322	201	191
1.5 to 3	340	234	183
1.5 to 4	356	261	185
1.5, no max.	368	278	184

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