

A Bioeconomic Analysis of Marine Reserves for Paua (Abalone) Management at Stewart Island, New Zealand

Viktoria Kahui · William Robert James Alexander

Accepted: 3 September 2007 / Published online: 25 September 2007
© Springer Science+Business Media B.V. 2007

Abstract Traditional fisheries management relies on the imposition of gear and/or effort restrictions. In spite of much experience with such techniques, the danger of fishery collapse is ever present. Biologists have advocated an alternative strategy, the establishment of marine reserves. However, it is possible that the benefits of marine reserve creation can be overstated if economic behaviour is ignored. In spite of being managed under a system of transferable quotas, the Stewart Island paua (abalone) fishery has been in decline for some time. We develop an integrated economic and biological model of this fishery and use it to predict biomass levels in a number of scenarios, including the imposition of a network of no-take areas. We identify circumstances under which the marine reserve solution outperforms traditional management techniques. We show that the benefit of a marine reserve is highest when a fishery is heavily exploited and when accounting for stochastic recruitment.

Keywords Bioeconomic modelling · Marine reserves · Paua (abalone) fisheries · Stochastic recruitment

1 Introduction

With commercially important fish stocks worldwide in decline despite intensive management efforts there is increasing pressure on governments to expand their marine conservation efforts (Allison et al. 1998), in particular to impose networks of marine reserves to act as hedges against failure of traditional management techniques and our uncertain knowledge of the marine environment. Fishery benefits of marine reserves are most commonly identified in terms of the spillover effects into adjacent fisheries (Rowley 1992) but also include protection against selective fishing and stock collapse from overexploitation. Although scientists have

V. Kahui
Department of Economics, NFH, University of Tromsø, Tromsø 9037 Norway

W. R. J. Alexander (✉)
Department of Economics, University of Otago, P.O. Box 56, Dunedin, New Zealand
e-mail: ralexander@business.otago.ac.nz

only limited experience with the effects of marine reserves (Ward et al. 2001), field studies do generally support the prediction that marine reserves increase biomass of fish stocks within the reserves, as well support larger average sizes of fish (Halpern 2003; Roberts and Polunin 1991, 1993; Mosquera et al. 2000; Ward et al. 2001). There are, however, only a few studies (Alcala and Russ 1990; Roberts et al. 2001) that directly support the notion of enhanced yields in areas adjacent to reserves. Roberts et al.'s (2001) study led to a number of comments criticising their findings (e.g., Tupper 2002 and Wickstrom 2002). Notably Hilborn (2002) questions the validity of observing increases in catch per unit of effort in absence of a reference site and in such a short timeframe. Smith et al. (2006) evaluate two recently implemented marine reserves in the Gulf of Mexico and observe a negative effect on catch displaying a downward trend.

In the absence of long-standing experience with the implementation of marine reserves, a number of biological studies have tried to simulate the effects of no-take areas. Such studies tend to ignore economic behaviour (Smith and Wilen 2003). The seminal model of Beverton and Holt (1957) demonstrated that marine reserves have to be very large to be on par with the yield gained under an optimally controlled fishery. Polacheck (1990) and DeMartini (1993) indicated a possible benefit of marine reserves from a shift in the age distribution within reserves. Inspired by the biological literature, a number of bioeconomic studies have emerged more recently. Holland and Brazee (1996), Hannesson (1998), Sanchirico and Wilen (1998), Conrad (1999), Pezzey et al. (2000) and Anderson (2002) generally take the view that payoffs from stand-alone marine reserves rarely compete with more traditional, optimal management schemes, but that they can be beneficial when stocks are heavily exploited. Sanchirico (2005) develops an analytically tractable bioeconomic model that allows comparisons of how differing assumptions on connectivity and dispersal affect conclusions on the net benefits of marine reserves but ignores such aspects as age and size specific effects as well as egg production per recruit. As Sanchirico (2005) notes, addressing such aspects requires the use of simulation methods. Spatial bioeconomic models, such as those of Holland (2000) and Smith and Wilen (2003), take this practical approach to integrating the biology and economics of marine reserves.

An increasing number of authors contend that while marine reserves might not be on par with the efficiency and harvest levels of a sustainable yield policy, their value lies in addressing the inherent uncertainty that pervades virtually every fishery (Guenette et al. 1998). For example, the collapse of cod stocks on Grand Banks, Canada, occurred despite best efforts by the Canadian government to apply world-class marine science and exploit stocks sustainably (Hannesson 1998). Lauck (1996) points out that in an ideal world, fisheries managers would have perfect knowledge of the marine environment and would be able to set adequate catch levels accurately. However, stock assessment models necessarily oversimplify the complex interaction between habitat/ecosystem quality, environmental changes, multi-species dynamics and recruitment variability when calculating a sustainable yield.

In this paper we focus on empirically-based simulations of the effects of a network of marine reserves in a specific paua (the indigenous name for abalone) fishery as an additional management tool to New Zealand's well-established quota system. Quotas are determined on the basis of a maximum sustainable yield, the calculation of which is based on the assumption of a homogeneously distributed stock. More sedentary species, such as abalone and oysters, however, display a distinctly patchy and heterogeneous pattern and are especially prone to serial depletion. We analyse catch and effort data from the abalone industry around Stewart Island, New Zealand, for the period between 1998 and 2003. A discrete choice model is applied to the data set, which is split into 16 distinct statistical areas. Simultaneous closures of selected areas are simulated according to the criteria of 'least economic impact' in terms

of the effect on redistribution and participation. A biological length-structured model for abalone is also established. The two models are then linked to gauge the effects in terms of biomass and economic impact of implementing a network of no-take areas.

Our research makes a valuable contribution to the literature in that we establish a bio-economic model based on real-life data and biological parameters incorporating both the predicted reduction and redistribution of effort, as well as stochastic recruitment, when evaluating the benefits of marine reserves. Some authors (Smith and Wilen 2003) take account of the redistribution of effort, while others (Conrad 1999) present analytical models that analyse marine reserve benefits by introducing a stochastic component. However, no study has yet incorporated both of these issues, and we show that for a heterogeneously abundant species, such as paua, spatial management in addition to quota limits could be vital in ensuring the long-term sustainability of the fishery.

This paper is organised as follows. Section 2 outlines some details of the Stewart Island abalone fishery and our data set. In Sect. 3 we develop an economic model of abalone diver behaviour. Section 4 sets out a model of the biology of the abalone species. Section 5 contains an explanation of how the economic and biological models are integrated and the results of the simulations. Section 6 concludes.

2 The Stewart Island Paua Fishery

Fisheries management analysis requires a good understanding of the biological characteristics of the fishery. We briefly outline the characteristics of paua and then summarise the raw data as provided by the New Zealand Ministry of Fisheries for this study.

Paua is among New Zealand's top 10 seafood export species and commercial catch has been divided into eight Quota Management Areas (QMAs) since the 1986–1987 fishing season (Annala et al. 2001), when it was adopted into the Individual Transferable Quota (ITQ) System. Anyone may enter the industry by buying quota at any time. There is, however, a maximum limit on quota holdings to inhibit monopolistic behaviour. Under the Fisheries Amendment Act 1986 no one can hold (own or lease) more than 20% of paua quota in a single QMA. Minimum holding limits for leasing and owning quota were also set in 1986. No person is able to hold quota equivalent to less than 1 tonne of paua within any QMA. The fishery is managed in conjunction with input based measures, such as minimum legal size and harvesting method restrictions. Paua are known to form large patches on reefs and movement of adult paua occurs over a very small spatial scale so that they are considered sedentary (Annala et al. 2001). However, paua are broadcast spawners and patches are loosely connected with each other so that local populations are aggregated into metapopulations (Shepherd and Brown 1993). Estimating paua abundance is difficult because divers tend to move amongst patches targeting large aggregations to maximise rent, and any significant changes in patch frequency over large areas of coastline (indicating relative abundance) are only expected to occur over long periods of time, even with heavy fishing, since catch quotas restrain harvest (McShane 1995). Even when depletion becomes apparent, average depletion might mask the disproportionate depletion of local stocks resulting in small-scale recruitment failure. Ministry of Fisheries' stock estimates determining yearly Total Allowable Commercial Catch (TACC) levels come with the caveat that serial depletion could cause model results to be overly optimistic. McShane et al. (1994) criticise current management efforts by the Ministry of Fisheries where quota recommendations are based on catch per unit of effort (CPUE) measures that treat a paua stock in any specific QMA as if it were a single stock with homogeneous biology, habitat and fishing pressure. They believe that

spatial management measures reflecting small local populations would be more appropriate. We chose the paua fishery in the QMA PAU 5B around Stewart Island because catch remains below the TACC (implying quota recommendations are too high) and has taken a marked decline since shortly before the turn of the century as well as the fact that catch and effort data are available from the Ministry of Fisheries for the period between 1998 and 2003 on a spatial scale fine enough to evaluate the imposition of no-take areas. The Ministry of Fisheries provided two raw datasets. The first one spans the time period from 01 January 1998 to 01 October 2001, where data are restricted to paua catch within the 16 statistical areas B1 to B16 of PAU 5B around Stewart Island. The second data set covers the time period 01 October 2001 to 05 August 2003, where catch is further subdivided into the new, smaller statistical area units of P5BS01 to P5BS84 and where data are recorded per diver. The relevant borders of the new 84 statistical areas coincide mostly with those of the original 16 areas, and all data from the 84 areas were summarised according to the 16 statistical areas because at that scale patches are large enough for travel distances between patches to be economically significant. Patches also have to be small enough to be subject to the same oceanographic currents but large enough to generate quantitative differences in expected rent. The size dimensions of the original 16 areas fulfil this criterion. Figure 1 shows the geographical locations of the 16 statistical areas in the QMA PAU 5B.¹ In merging the two datasets, a number of adjustments had to be made including aligning identification information to account for changes in reporting requirements over the period under consideration and removing a small amount of anomalous data. The final combined dataset contains 3,388 observations on trips made to any particular patch on any day between 01 January 1998 and 01 July 2003. Paua divers fill in a daily form providing information on the fishing date, method of catch, statistical area of catch, total gathering and/or diving time, number of people gathering or diving during the day, target species and the estimated weight of catch (kg).²

3 Modelling Diver Effort

We now outline how a discrete choice model, and the nested logit model in particular, lends itself to application to our paua data set. We fit such a model and use it to simulate the simultaneous closure of selected areas according to the criterion of ‘least economic impact’ in terms of the effect on redistribution and participation.

Bockstael and Opaluch’s (1983) work on supply response decisions by fishing firms was the first to use a multinomial logit approach to deal with the behavioural motivations of fishermen. Here we follow the modelling approach developed in Smith and Wilen (2003). We postulate that the fishermen’s problem is to select from a computationally manageable number of fishing areas according to an index of attractiveness. The attractiveness of each fishing area is defined by its profit, which is a function of catch, prices and costs that depend on travel time and boat characteristics. Fishermen are assumed to maximise profits and will make the location choice that will yield the highest expected utility.

¹ Note paua is harvested on rocky habitats to a depth of 10–15 m, thus the actual harvestable area is unlikely to be directly proportional to the size of the statistical area. One can imagine a continuous reef tracing the outline of Stewart Island with clusters of paua scattered along it. While it is difficult to make any precise prediction about how these clusters compare in each statistical area in terms of size, it is reasonable to assume that these areas have been designed to more or less evenly split the overall area into harvesting zones.

² The data provided by the Ministry of Fisheries are commercially sensitive, i.e. there is no way to link information on individual divers’ traits with the diving occasions recorded.

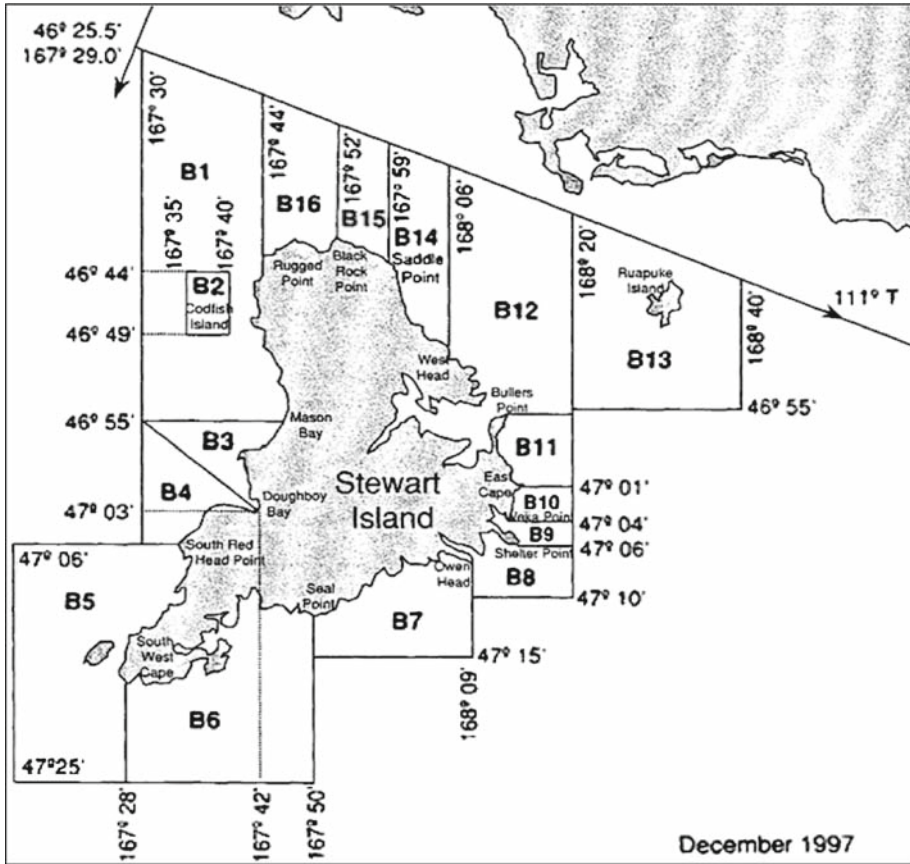


Fig. 1 Statistical areas B1–B16 for PAU 5B

We assume that on any given day between 01 January 1998 and 01 July 2003 paua divers choose whether to go diving or not. This choice will depend on a variety of factors, such as weather conditions and expected abundance levels.³ Once the decision to go diving has been made, divers choose among the 16 patches in PAU 5B based on spatially varying abundance levels and travel costs. These two decision nodes can be econometrically addressed by a nested logit model (Smith and Wilen 2003). The NL model avoids the assumption of independence of irrelevant alternatives (IIA), which implies that the odds of visiting area j instead of not diving does not depend on the number of other sites available, their characteristics and their costs. It would thus predict a uniform participation response to closure, i.e., effort is uniformly distributed and the imposition of a no-take area leads to a proportionate decrease in effort. The NL model also allows us to evaluate the elimination of specific diving patches and to predict subsequent changes in participation decisions and redistribution of trips.

McFadden (1973) has derived an explicit representation of the probabilities given the assumption that the errors are independent and identically distributed (i.i.d.) with a Weibull

³ Note that due to the small size of the island and the nature of the fishery, boat sizes are fairly homogeneous in the industry and are privately owned (personal communication with divers/observation). The configuration of the data entries show that these boats typically hold 5–15 people at a time.

(extreme value) distribution. The NL model then explicitly states the probability of choosing area j in the lower branch l of the decision 'DIVE' as

$$P_{j|l} = \frac{e^{\beta' X_{j|l}}}{\sum_{j=1}^{J_l} e^{\beta' X_{j|l}}} \quad P_l = \frac{e^{\gamma' Z_l + \tau_l I_l}}{\sum_{l=1}^L e^{\gamma' Z_l + \tau_l I_l}} \quad (1)$$

$X_{j|l}$ contains the location-specific attributes of expected catch per unit of effort (CPUE)⁴ and travel distances to the various patches from the main port in Halfmoon Bay (DISTANCE).⁵ Z_l describes the attributes of the choice sets, i.e., 'Not Dive' and 'Dive'. These include daily observations on selected indicators of the weather conditions.⁶ In addition, a seasonal dummy is included indicating the winter season in the southern hemisphere from 01 March to 01 October (DSEASON).⁷

The discrete choice of diving location serves as the dependent variable. Thus, the duration (i.e., hours dived) at any patch has to be highly correlated to the number of trips taken in order to produce consistent results. A simple Ordinary Least Square regression of duration on trips reveals a highly significant coefficient (5.387) with a p -value < 0.0005 and a R-square adjusted value of 0.9712. The parameters of the NL model were estimated for the time period 01 January 1998 to 01 July 2003 via the full information maximum likelihood (FIML) estimation with NLOGIT Version 3.0. The utility of 'not diving' was normalized to zero.⁸ After rejecting the IIA,⁹ the NL model was estimated and Table 1 shows the results. The top part of the table shows the proportions of choices made at each decision node. Divers decided to go diving nearly 75% of the time. Area 12 received the largest proportions of visits (13%), closely followed by Area 14 (9%) and Area 2 (7%). The spatial harvesting behaviour

⁴ Ideally, we would want to use patch-specific expected revenue and expected costs as explanatory variables. However, pua prices are not readily available and we use a monthly backward looking average of pua catch per unit of effort for each statistical area as an indicator of profitability. The bulk of pua is destined for overseas export and is sold in a processed state (SeaFIC 2003), which suggests a less volatile price than one would expect from overnight live seafood freight, such as rock lobster, to Asian markets. Furthermore, producers do not pay different quality premiums by area or season due to the fact that all commercial pua is sold directly to processing plants where it is canned (personal communication with divers). More recently we managed to obtain yearly port prices for PAU 5B between 2003 and 2006. Prices vary slightly from year to year between NZ \$37–43 per kg.

⁵ We constructed a distance variable as a proxy to the costs pua divers incur when choosing a statistical area. Distances are measured on the map (see Fig. 1) between the midpoints of the statistical areas tracing the coastal outline of the island. The 'switching point' lies between area B4 and B5 (on the opposite side of the island from Oban), i.e., it is cheaper (in terms of distance travelled) to approach area B4 heading around the north of the island, but B5 around the south of the island. Pua divers who fish in PAU 5B typically live in Oban at Halfmoon Bay since it is the only settlement providing accommodation, storage and safe anchoring. Multi-day trips are rare since the island can be navigated within a few hours and divers prefer to return to Oban by the end of the day (personal communication). Similarly, there were only very few multi-area dives, mostly to areas 3, 4 and 5, in which case distances for the particular trip were adjusted accordingly.

⁶ We have daily observations from 01 January 1990 to 01 July 2003 on: RAIN (total rainfall in millimetres); MAX (maximum temperature in °C); MEAN (mean temperature in °C); MIN (minimum temperature in °C); MSLP (daily mean of mean sea level pressure in hectopascals); WIND (mean wind speed in metres per second); and GUST (maximum gust speed in metres per second).

⁷ The data were checked for the 'weekend effect' but no significant difference was found with respect to the day of the week chosen for diving. The pua diving industry is very informal and known to be a lifestyle choice rather than a conventional job.

⁸ The utility of 'not diving' can, in fact, be normalized to any number as long as the remaining utilities are scaled correspondingly.

⁹ The Hausman and McFadden (1984) specification test for the multinomial logit model has been used to test the hypothesis of the IIA.

Table 1 Nested Logit Model estimation results

Variable	Coefficient	Standard Error	b/St.Er.	P[Z > z]
Maximum likelihood estimates				
Log likelihood function <i>LU</i>	-8369.574		Chi squared	1955.031
Restricted log likelihood <i>LR</i>	-9347.090		Pr[chi-sqrd > value] =	.0000
Rsqrd	.10458		RsqAdj	.10441
CPUE	.0104	.0005	19.921	.0000
DISTANCE	-.1122	.0080	-14.005	.0000
DIVE	.8991	.3640	2.470	.0135
NOTDIVE	1.0000	(Fixed Parameter)		
Constant	-4.3695	5.2331	-.835	.4037
RAIN	-.0126	.0082	-1.538	.1241
WIND	-.2299	.0437	-5.258	.0000
MSLP	.0051	.0051	.992	.3211
GUST	-.0650	.0191	-3.409	.0007
MEAN	.0403	.0136	2.955	.0031
DSEASON	-.8941	.1090	-8.203	.0000

of divers supports the biological finding that paua tends to aggregate in clusters with different levels of abundance.

The lower part of the table reports the coefficients of the estimating equation. Most are statistically significant at the 5% level. Although the coefficients are not directly interpretable, as they would be for a linear model, they do provide a general indication of the direction of effects. The coefficients of the attributes CPUE and DISTANCE in the location-specific utility functions are highly significant and support the intuition that patch-specific catch per unit of effort measures impact positively on the choice of a specific patch, while the travelling distance has a negative impact. The coefficients of the attributes explaining the choice between 'Dive' and 'Not Dive' also exhibit the correct signs: the probability of going diving on any given choice occasion is negatively impacted by RAIN, WIND, GUST and the dummy variable DSEASON, which indicates winter time. MEAN (temperature) has a positive impact. The constant term, RAIN and MSLP are insignificant at 5% significance level, but decrease the fit of the model when left out. Rain does not influence the riskiness or success of diving, and is also not necessarily an indicator of bad weather. High and low pressure are usually associated with good and bad weather, respectively, but we suspect that the stationary measure of the MSLP at the weather station does not adequately reflect the pressure gradient. Strong pressure gradients, which are identified by a tight packing of isobars, are associated with stronger winds, and the estimates of WIND and GUST show that it is the level of wind that has a significant influence on the decision to go diving.

Table 1 also reports the likelihood ratio test where the log-likelihood functions are evaluated at the unrestricted ($LU = -8369.574$) and restricted ($LR = -9347.090$) estimates. The value of the chi-squared statistic exceeds the critical value and we reject the joint hypothesis that the coefficients on the 10 explanatory variables are all zero ($< .00005$). The measure of goodness-of-fit for discrete choice models is analogous to the conventional R^2 calculated as a likelihood ratio index $LRI = 1 - LU/LR$. While it is common for researchers to report this measure, it can not be used to suggest how well the model predicts the outcome variable.

Similarly, the conventional $R^2=0.10458$ and adjusted $R^2=0.10441$ values are not indicative of the fit of the model. Intuitively, these measures are strongly linked with the likelihood ratio test and they might give some indication of the improvement that the unrestricted model gives over the restricted one, thus the focus should really be on the likelihood ratio test.

We reject the null hypothesis that the coefficient of the inclusive value parameter (IV) for the branch 'DIVE' is equal to zero, i.e., at 5% significance level DIVE is significantly different from zero. The positive sign of the coefficient indicates the positive expected utility divers gain from participation. The IV for the coefficient 'NOT DIVE' has been fixed at 1 because the nesting structure has a degenerate branch (Hensher and Greene 2000). The statistical significance of the IV for DIVE gives confidence in the correct specification of the model.

As the coefficients of the nested logit model are not directly interpretable, the economic effects of the coefficients have to be derived from the elasticities of the probabilities. Calculated own and cross elasticities indicate that the own elasticity of each area is positive and relatively inelastic for the attribute CPUE, while negative cross elasticities show that diving patches are close substitutes for each other. We also find that patches 3, 4 and 5 are elastic to changes in distances.

In order to simulate the closure of a selected no-take area the computed probability of visiting that patch has to be extremely small. One can achieve this by setting the CPUE of the area in question to a very large negative number (such as $-1,000$) or the distance to a very large positive number (1,000). This has the effect of reducing the indirect utility of the selected area to a very large negative number resulting in the prediction that no trips are taken to this area (Smith and Wilen 2003). To yield the management benefits of marine reserves, such as insurance against environmental and management failure as well as a sustainable means of stock enhancement, we need to simulate the closure of a number of patches simultaneously. Christie et al. (2002) suggest that a network of no-take areas nested within a broader management strategy to restrict effort is preferred in terms of abundance in areas adjacent to the no-take areas than a single, isolated marine reserve. Experimental simulations, which are short-term and not bioeconomic at this stage, have been performed for closure of 4 statistical areas at once,¹⁰ where closures of high CPUE with low CPUE areas are compared, as well as combinations of both extremes. Areas 3, 4 and 5 distinguish themselves by yielding the smallest redistribution and participation effects. In particular, the smallest percentage change in the number of trips taken to Areas 4 (-2.34%) concurs with the observation that Area 4 has the smallest proportion of visits, the smallest levels of CPUE and yields a minor redistributive impact on effort. Preliminary data analysis shows Areas 4 and 5 to be frequently targeted as part of one trip, i.e., they are the only areas elastic with respect to distance (see footnote 5). Areas 4 and 5 also measure the furthest distance from the port, and we decide to use these areas as starting point, where various other areas are added individually according to the criteria of percentage changes in the number of trips and levels of cross elasticity. These values have direct implications for the redistribution of effort and the participation decision. In addition, the geographical positioning of such no-take areas is important. The selected no-take areas should be spaced out as much as possible in order to achieve the desired 'network' effect.

The best results with respect to participation and distribution impacts are achieved for closures of areas 4, 5, 13, and 16 (areas 4, 5, 13, 1) and areas 4, 5, 9 and 16 (areas 4, 5,

¹⁰ Given 16 statistical areas, it seemed reasonable to start with a modest number of 4 no-take areas. However, any number of closures can be simulated and with the addition of biological information, the choice of how many areas should be closed will become less discretionary.

Table 2 Simulation results for a network of no-take areas

	Baseline prediction	4, 5, 13, 16 closed	4, 5, 13, 1 closed	4, 5, 9, 16 closed	4, 5, 9, 1 closed
	% Share	% Change	% Change	% Change	% Change
NOTGO	21.30	3.24	3.29	3.32	3.37
AREA1	4.58	0.93	-4.58	0.94	-4.58
AREA2	4.26	0.87	0.89	0.89	0.91
AREA3	3.45	0.73	0.73	0.73	0.74
AREA4	2.34	-2.34	-2.34	-2.34	-2.34
AREA5	3.76	-3.76	-3.76	-3.76	-3.76
AREA6	4.30	0.87	0.88	0.86	0.87
AREA7	5.37	1.06	1.08	1.07	1.09
AREA8	5.70	1.17	1.20	1.19	1.22
AREA9	5.70	1.16	1.17	-5.70	-5.70
AREA10	6.00	1.22	1.24	1.24	1.25
AREA11	6.33	1.30	1.31	1.32	1.33
AREA12	6.52	1.34	1.35	1.35	1.36
AREA13	5.52	-5.52	-5.52	1.11	1.13
AREA14	5.52	1.12	1.13	1.13	1.14
AREA15	4.96	1.02	1.03	1.04	1.05
AREA16	4.39	-4.39	0.90	-4.39	0.92
Total	100.00	0.00	0.00	0.00	0.00

9, 1).¹¹ Table 2 shows that a network of such no-take areas around Stewart Island would lead to a reduction in participation (i.e., an increase in 'NOTGO') by only 109–114 trips between 1998 and 2003, i.e., by 3.24–3.32%. Effort tends to redistribute mainly to the Areas 9, 10, 11 and 12, areas which are characterised by high relative catch rates. This redistribution effect will put more pressure on existing 'fishable' areas and might lead to significant decreases in existing location specific CPUE over time. This effect is not captured in the simulations and, undoubtedly, displaced effort might outweigh any stock enhancement benefits gained; however, such conclusions have to incorporate biological and economic information jointly. In the absence of any biological input, the economic data suggest that imposing no-take areas according to 'least-impact' criteria could be a viable option without leading to substantial economic losses even though actual costs to the fishery cannot be measured in the absence of information on expected revenues, i.e., the criterion of least economic cost simply allows us to rank the options in terms of loss of diving trips undertaken, and therefore the loss of income from reduced harvest opportunities. Further work can include estimates on short-term and long-term welfare losses (see for example Hicks et al. 2004), but this is beyond the scope of this paper.

¹¹ Areas 1 and 16 can be closed interchangeably, i.e., they are adjacent to each other and yield the same quantitative results.

4 Biological Model

In order to simulate biomass changes of the paua population around Stewart Island, we use a length-class, or Beverton–Holt model. Biological estimates for many of the crucial parameters for such a model are known for the paua fishery around Stewart Island. The parameters and data used here apply specifically to the paua stock in PAU5B. [Breen et al. \(2000\)](#), who were commissioned by the Ministry of Fisheries, employ Bayesian estimation techniques to provide quota recommendations. We incorporate Breen et al.’s (2000) equations on paua growth, survival, recruitment, fishing mortality and the corresponding biological parameters while information on egg production and fecundity is drawn from other sources of research on paua. We extend Breen et al.’s (2000) work by applying area-specific catch levels provided by the dataset used for the discrete choice analysis to determine area-specific levels of spawning biomass, egg production, legal fishing biomass and numbers of paua. These values are then used as starting points for the ensuing area-specific bioeconomic analysis.

4.1 Paua Growth

[Breen et al. \(2000\)](#) divide length of abalone into 50 ‘bins’, each advanced by a 2 mm step in shell length for a range of 70–168 mm. Recruitment enters the first bin. The last bin contains all individuals 168 mm or more in length. Growth is determined for each length class according to the von Bertalanffy growth Eq. (2). The expected average yearly growth increment is calculated for each length class according to

$$\Delta l_k = (L_\infty - l_k) \left(1 - e^{-K}\right) \quad (2)$$

where l_k represents the average shell length of paua in bin k , L_∞ the maximum possible size of paua and K the Brody growth coefficient. The calculated growth increments are assumed the same for all of the 16 statistical areas in PAU 5B.¹² With a given standard deviation of growth increments around the mean, $\sigma^{\Delta l_k}$, the distribution of increments for each length class k can be calculated from the normal distribution. Mean growth increments for abalone in higher length classes are smaller and any negative values are calibrated to zero implying no growth. Ideally a non-constant standard deviation should be applied but little biological information is available. It is then possible to estimate a vector of probabilities with respect to the transition of abalone moving from one length class to any of the larger length classes, with all length classes combined forming a transition matrix G . All probabilities in G add up to one, and the probability of remaining within the last bin is set equal to one to bound growth to the largest size class. Preliminary sensitivity trials with different standard deviations around the growth mean and different calibrations of negative values show some impact on legal biomass estimates, especially on paua in the larger length classes due to the exponential weight-at-length relationship. However, the impact is small enough to warrant the same qualitative conclusions to be drawn in the ensuing bioeconomic simulations. To be consistent with the stock assessment results provided by the [Breen et al. \(2000\)](#) we adopt their base case (BC) values ($L_\infty = 150$ mm, $K = 0.252$, $\sigma^{\Delta l_k} = 4$ mm) but we also apply their suggested lower (LO) ($L_\infty = 140$ mm, $K = 0.343$, $\sigma^{\Delta l_k} = 4$ mm) and a higher value

¹² In practice there will be some variation in growth parameters for different areas around Stewart Island. However, area-specific estimates do not exist apart from a few selected sites. [Schiel and Breen \(1991\)](#) estimated von Bertalanffy growth parameters at Stewart Island west, north and east. Their estimates are close enough to assume the same L_∞ and K for the whole area of PAU 5B.

(HI) ($L_\infty = 160\text{ mm}$, $K = 0.198$, $\sigma^{\Delta k} = 4\text{ mm}$) in order to gauge the sensitivity of the results to the parameters used.¹³

4.2 Survival

The biological model has to be ‘burnt in’ for a length of time with all reproductive parameters to allow numbers-at-length $N_{k,t}$ to reach a steady-state prior to when fishing started in 1974. In each of these years

$$N_{1,t} = R0 = \exp(Rc\text{off}) \tag{3}$$

where $Rc\text{off}$ represents the natural logarithm of $R0$, the average number of recruits to the virgin population.¹⁴ Yearly abundance values (i.e., numbers-at-length) are advanced according to

$$N_t = N'_{t-1} \bullet G \exp(-M) \tag{4}$$

N_t is a matrix showing the number of paua in each of the 50 length bins and M represents the instantaneous natural rate of mortality.

4.3 Recruitment

The survival rate from egg to adult abalone is thought to be extremely small. Larvae are very hard to locate during field samples as they are very patchy, thus the larval stages are the least understood stages of abalone recruitment. Breen et al. (2000) assume a Beverton–Holt recruitment relationship and calculate the two parameters from spawning biomass prior to the onset of fishing, $B0^{Spawn}$, equilibrium recruitment, $R0$, and steepness,¹⁵ h . Yearly spawning biomass is determined by

$$B_t^{Spawn} = \sum N_{k,t} P M_{k,t} w_k \tag{5}$$

where w_k represents the mean weight and $P M_{k,t}$ is a switch (1=on, 0=off) that indicates whether length bin k is above the length at which individuals become mature in year t . The Beverton–Holt relationship is determined as

$$\alpha = \left(B0^{Spawn} / R0 \right) (1 - (h - 0.2) / (0.8h)) \tag{6}$$

$$\beta = (h - 0.2) / (0.8h R0) \tag{7}$$

$$R = \frac{B^{Spawn}}{\alpha + \beta B^{Spawn}} \tag{8}$$

where $\alpha = 1/a$ (a represents the maximum number of recruits per spawner at low stock sizes, i.e., it captures the resilience of paua recruits), and $\beta = 1/b$ (b indicates the maximum number of recruits when the spawning stock is very large, i.e., it is a carrying capacity parameter). The understanding of the relationship between spawning biomass and subsequent recruitment is the most crucial issue in identifying population dynamics (Myers 2001). The parameter a

¹³ We generated different sets of growth matrices but our virgin biomass estimates always differed to Breen et al.’s (2000a) estimates by roughly 20%. Personal communication with the authors revealed differences in assumptions about the paua in the last few length bins, which explained the discrepancies in estimates. The authors kindly provided us with their growth matrices to ensure consistency in results.

¹⁴ The virgin population is defined as the original level of population prior to any fishing activities.

¹⁵ The percentage of eggs that will be produced when spawning biomass is at 20% of its virgin level is expressed by h , the ‘steepness’ of the relationship.

gives the slope of the function at $B_t^{Spawn} = 0$, which can be interpreted as the maximum annual reproductive rate, which is important to estimating the population growth rate.

4.4 Egg Production

Fecundity is approximately proportional to body weight. Older females produce far more eggs than newly matured females. Abalone populations often contain accumulations of large, old individuals who contribute most of the reproductive effort to the population. Length-fecundity relationships can be found in Poore (1973) and Sainsbury (1982). The annual number of eggs produced can be calculated similarly to the spawning biomass

$$N_t^{Eggs} = \sum N_{k,t} P M_{k,t} E_k \tag{9}$$

The total number of eggs N_t^{Eggs} is determined by the number of eggs E_k produced at each specific length class according to a length-fecundity relationship summed over all bins k .

4.5 Fishing Mortality

Yearly legal biomass levels are determined according to the equation

$$B_t^{legal} = \sum N_{k,t} P_{k,t} w_k \tag{10}$$

Exploitation rate U_t is then found by model biomass and observed catch C_t

$$U_t = C_t / B_t^{legal} \tag{11}$$

Survival from fishing $SF_{k,t}$ is calculated as

$$SF_{k,t} = 1 - U_t P_{k,t} \tag{12}$$

The vector of numbers-at-length from 1974 onwards, when commercial fishing in PAU 5B started, is advanced each year by the current vector of numbers-at-length, the vector of survival from fishing, the growth transition matrix, natural mortality and the number of recruits determined by the Beverton–Holt relationship in Eq. (8) entering the first bin in each year.

$$N_t = SF_t N'_{t-1} G \bullet \exp(-M) \tag{13}$$

4.6 Biological Parameters

Growth rates are probably the most extensively studied aspect of pua biology (for example Poore 1973; Sainsbury 1982; Schiel and Breen 1991). Also, morphology, reproductive systems, breeding seasons and the relationship between age, fecundity and minimum size at first maturity are known relatively well for many genus *Haliotis* species. However, natural mortality and the relationship between standing stock and recruitment of pua are still poorly understood. Table 3 lists a realistic set of values provided by Breen et al.’s (2000) sensitivity trials for natural mortality (M) and the two recruitment parameters (R_{coeff} and h). M and R_{coeff} are highly correlated, therefore a change in one of these values implies a change in the other. All parameters in Table 3 apply equally to all 16 statistical areas within PAU 5B. Only the calculated Beverton–Holt parameter β is adjusted to reflect the size of each of the statistical areas according to the historic percentage catch.

Table 3 Biological parameter values

Parameter	Description	Value	Source
w_k	Weight-at-length (g)	$w_k = (2.99 \times 10^{-5}) \times \text{length}^{3.303}$	Schiel and Breen (1991)
$PM_{k,t}$	Sexual maturity	'switch' at 92 mm	Breen et al. (2000)
$P_{k,t}$	Minimum legal size	'switch' at 126 mm	Breen et al. (2000)
E_k	length-fecundity	$E_k = (9.32 \times 10^{-12}) \times \text{Length}^{8.408}$	Schiel and Breen (1991)
M	Natural mortality	0.108; 0.12; 0.132;	Breen et al. (2000)
R_{coeff}	Recruitment	13.971; 13.67; 14.37	Breen et al. (2000)
h	Steepness	0.4; 0.21; 0.7	Breen et al. (2000)

4.7 Model Dynamics for the Whole of PAU 5B

Having established the algebraic form of iteration and a possible set of biological parameters, we lay out the process of finding the legal virgin biomass level as of 1974, just before commercial harvest of paua commenced, as well as the legal biomass level as of 1997, just before the Ministry of Fisheries started recording catch data on an area-specific level.

Paua abundance levels are 'burnt in' for 60 years according to Eqs. (3) and (4), after which they converge. Table A1 in the Appendix shows the sensitivity of biomass estimates to changes in the natural mortality rate, M , for the three different growth transition matrices BC, LO and HI. The values in bold in Table A1 shows the sensitivity trials performed by Breen et al. (2000), which we expand on by applying the different M values more systematically to BC, LO and HI. In the base case BC Breen et al.'s (2000) assume $M = 0.12$ in combination with $R0 = 1, 168, 230$. For this scenario, we estimate a virgin legal biomass of 2,496t, which lies 3t below Breen et al.'s (2000) estimate. For LO Breen et al. (2000) assume $M = 0.132$ in combination with $R0 = 1, 741, 052$. Due to observed biological relationships, different assumptions for L_∞ and K imply a different value for M . Similarly, M and $R0$ are highly correlated and are therefore varied together. Again, our estimated virgin legal biomass level for LO of 2,914t is very close to Breen et al.'s (2000) result (2,950t). The same applies to HI for $M = 0.108$ and $R0 = 864, 581$, where our model estimates 2,327t and Breen et al. (2000) find 2,320t. Within each scenario, increasing the value of M from 0.108 to 0.12 and to 0.132 decreases the overall virgin legal biomass from, for example, 3,004t to 2,496t and 2,099t in the base case BC. The same trend is observed for LO and HI. Overall, the model behaves as we would expect and our virgin legal and spawning biomass estimates fit comfortably with estimates by Breen et al. (2000), which provides confidence in the accuracy of the biological model. We conduct further sensitivity trials to changes in h and R_{coeff} . Overall, sensitivity trials demonstrate the model to be relatively sensitive to changes in M and R_{coeff} , but differences in biomass estimates remain within a range reasonable enough to warrant the same qualitative conclusions to be drawn in future simulations.

Having identified virgin abundance and biomass levels, we now advance yearly stock estimates under fishing pressure according to the growth transition matrix BC, natural mortality and survival. Recruitment is not assumed constant any more but is determined by the Beverton–Holt equation where recruitment enters the first length bin each year. Total numbers of paua in each length bin of the virgin biomass are extracted and used as starting values. We simulate biomass levels up to the year when catch was first recorded by statistical area, i.e., in 1998. Between 1974 and 1997 catch was recorded for the *overall* quota management

area PAU 5B by the Ministry of Fisheries, and we add the 23-year base case catch trajectory provided by [Breen et al. \(2000\)](#) to our model. In [Table A1](#) the virgin biomass estimate for BC implied a value of 957 for α . However, when plugging this value into the Beverton–Holt recruitment function, the legal biomass level drops down quite significantly to 231t in 1997, and collapses a few years later. Random trials with a lower value for α , i.e., a higher annual maximum reproductive rate, yield an estimated legal biomass that matches [Breen et al.’s \(2000\)](#) biomass estimate in 1999 of 637t and we anchor the model at $\alpha = 655$. The reason for the discrepancy in the α value lies in the recruitment function (personal communication with Paul A. Breen), i.e., [Breen et al. \(2000\)](#) add a stochastic component to the recruitment function after virgin biomass levels are identified. In order to calibrate the model to reflect the biomass levels identified by the [Breen et al. \(2000\)](#) as of 1997, we assume a deterministic recruitment function at this stage and consider the effects of introducing the stochastic component in the ensuing bioeconomic simulations.

4.8 Area-Specific Model Dynamics

We now extend the analysis by [Breen et al. \(2000\)](#) by focusing on the dynamics in each of the 16 statistical areas rather than just the whole of PAU 5B. The ‘area-specific catch dataset’ used for the discrete choice model provides area-specific levels of catch between 1998 and 2002 (data are only partially available for 2003). We estimate legal biomass levels in tonnes, B_{Legal} , for the whole of PAU 5B as well as for the individual statistical areas between 1998 and 2002. The overall legal biomass level drops from 666t in 1997 to 636t in 2000, but picks up again in 2001. The highest levels of fishable biomass are found in areas 2, 12 and 14. The length-based model also estimates area-specific egg production, exploitation rates, recruits and total numbers of paua.

5 A Bioeconomic Simulation Model

The economic and biological models can now be integrated, but first, we evaluate the effects of a network of marine reserves by simulating legal biomass up to 100 years ahead (but focusing on the next 10 years) without any closure. We then impose closed areas but without considering the redistribution of effort predicted by the economic model and finally we allow the economic response from divers. The comparison of legal biomass levels in these three scenarios enables us to draw some conclusions about the potential benefits of marine reserves. We then re-evaluate the conclusions drawn by investigating the effects of spillover gradients and allowing for a stochastic component in the recruitment function to take account of the variability of the marine environment.

5.1 Without Closure

The paua fishery in PAU 5B has undergone significant reductions in TACC and harvest levels since 1995. Previously, we argued that between 1995 and 2001 the PAU 5B fishery showed symptoms of unsustainable harvest as catch has remained consistently below the TACC since the turn of the century. During this time, the TACC had been slowly reduced from 148.98t in 1996 to 112.19t in 2002. The most recent stock assessment of PAU 5B on behalf of the Ministry of Fisheries ([Annala et al. 2005](#)) shows that the TACC has been further decreased from 112.19t in 2002 to 90t in 2003 and 2004. Since then, catch has remained stable and close to the TACC, which implies that the current TACC could possibly be a sustainable yield

Table 4 Constant TACC

	2003	2005	2007	2009	2011	2013	2113
C (t)	90	90	90	90	90	90	90
BLegal (t)	661	688	710	731	752	774	1,527
BSpawn (t)	1,013	1,039	1,065	1,090	1,116	1,143	2,010
R	846,100	858,137	869,302	880,268	891,214	902,157	1,161,631
TEgg $\times 10^{+13}$	2.429	2.549	2.656	2.757	2.854	2.951	6.421
TN (millions)	4.475	4.544	4.621	4.702	4.786	4.872	7.354
U (%)	13.6	13.1	12.7	12.3	12.0	11.6	5.9

and might not have to be reduced further. We can test this assumption by adding a constant yearly harvest rate equal to the current TACC (90t) to our biological model.¹⁶ We extract numbers-at-length of paua as of 2002 in each statistical area and use them as starting values for the simulations in the following scenarios.

In each year, legal biomass (BLegal), spawning biomass (BSpawn), number of recruits (R), levels of egg production (TEgg), total number of paua (TN) and exploitation rates (U) are simulated through constant fishing, growth and natural mortality. Assuming constant catch and constant distribution of effort (according to the historic percentage catch of the area-specific catch dataset used for the discrete choice analysis), Table 4 shows that the current TACC for paua around Stewart Island is currently set at a sustainable level. Overall legal biomass levels¹⁷ are predicted to increase from 661t in 2003 to 774t in 2013, and approach an equilibrium of 1,527t one hundred years later. Our findings are supported by Annala et al.'s (2005) recommendation that the current TACC of 90t as of 2004 will likely allow the stock in PAU 5B to recover. Note that although our results are not directly comparable due to different biological base case assumptions, our model behaves in a way which allows the same qualitative conclusions to be drawn, i.e., biomass levels are predicted to have decreased up to the point when the TACC was lowered to 112t and then 90t, after which biomass levels are predicted to rise again. This provides confidence in the ability of our model to evaluate the relative effects of imposing a network of marine reserves in reference to estimates by Breen et al. (2000). Further simulations adding various constant harvest rates between 90t and 110t show that a constant catch rate of 105t tips the balance where biomass levels drop and eventually collapse.

It might be more reasonable to assume that the Ministry of Fisheries will raise TACC levels as biomass levels increase over time. Table 5 shows increasing levels of catch in response to increasing levels of biomass over time. With the current TACC of 90t as a starting point, we keep the corresponding 2003 exploitation rate U in each statistical area fixed (ranging from 12.5 to 14.9%). The simulation results in Table 5 show that only modest increases in catch reaching an upper limit of 106t in 2113 are attainable when assuming a fixed exploitation rate. The assumption of a fixed exploitation rate might be reasonable given regular paua sample surveys undertaken, which are incorporated in Ministry of Fisheries stock assessments.

¹⁶ We refer to Schaefer's model of stock density-dependent growth in terms of its stable and unstable equilibrium. If biomass levels converge to the stable equilibrium when adding a constant level of harvest to the biological model, we can deduce that the relevant harvest level is sustainable.

¹⁷ Our model generates area-specific biomass levels but at this stage we focus on the overall performance of PAU 5B.

Table 5 Fixed exploitation rate

	2003	2005	2007	2009	2011	2013	2113
C (t)	90	94	96	97	98	99	106
BLegal (t)	661	686	701	712	721	730	782
BSpawn (t)	1,013	1,037	1,055	1,070	1,083	1,094	1,161
R	846,100	857,181	865,069	871,510	877,004	881,707	909,290
TEgg $\times 10^{+13}$	2.429	2.541	2.621	2.681	2.729	2.768	2.986
TN (millions)	4.475	4.530	4.581	4.627	4.668	4.704	4.920
U (%)	13.6	13.6	13.6	13.6	13.6	13.6	13.6

Table 6 Closure of areas 4, 5, 9 and 16 without economic model

	2003	2005	2007	2009	2011	2013	2113
C (t)	79	82	84	85	86	87	94
BLegal (t)	661	707	740	768	795	820	1,050
BLegal fishable (t)	582	604	617	627	636	643	689
BSpawn (t)	1,013	1,058	1,096	1,131	1,163	1,194	1,453
R	846,100	865,920	881,006	893,566	904,326	913,568	968,569
TEgg $\times 10^{+13}$	2.429	2.615	2.775	2.917	3.047	3.170	4.354
TN (millions)	4.502	4.614	4.723	4.826	4.921	5.008	5.653
U (%)	13.6	13.6	13.6	13.6	13.6	13.6	13.6

However, the lack of a large-scale assessment of paua abundance coupled with administrative lags in quota recommendations warrant some caution. We use both the assumption of constant catch and fixed exploitation rate in our simulations as discussed below.

5.2 Closure but no Redistribution of Effort

We ‘close’ areas 4, 5, 9 and 16 to fishing by setting catch equal to zero in these areas whilst anchoring catch at the 2003 exploitation rate in each of the remaining statistical areas. Table 6 shows the resulting biomass levels a model without economic response would predict. Initially, catch drops from 90t to 79t. Spawning biomass increases at a faster rate than under no closure (see Table 5) and egg production increases exponentially to 4.354×10^{13} in 2113. The total legal biomass of 1,050t in 2113 is significantly above the legal biomass predicted in Table 5 (BLegal=782t), but fishermen and policy makers would most likely be interested in levels of ‘fishable legal biomass’ rather than total legal biomass levels.¹⁸ As expected, fishable legal biomass in Table 6 (BLegal fishable=689t) lies below legal biomass in Table 5 (BLegal=782t) since closure of areas 4, 5, 9 and 16 implies less area available for fishing.

¹⁸ Fishable legal biomass is obtained by summing the legal biomass in areas 4, 5, 9 and 16 and subtracting it from the overall legal biomass in each year.

5.3 Closure with Redistribution of Effort

Before combining the economic with the biological model, we have to translate the change in number of trips predicted by the nested logit model (see Table 2) into a change in predicted catch when imposing a network of no-take areas. This is so because the amount of catch, rather than the number of trips, is fed into the biological model for each of the statistical areas in PAU 5B to reflect its effect on stock levels. Exploitation rates are then calculated according to Eq. (11). The results of a simple ordinary least squares regression of catch on trips (using the area-specific data set) indicates that the intercept is significant at the 10% significance level ($p=0.0931$) and the coefficient for TRIPS is highly significant at the 1% significance level ($p < 0.0001$). The F -statistic for the overall model is highly significant ($F=282.45$, $p < 0.0001$), indicating that the model explains a significant proportion of the variation in the data. R-Square (0.815) and Adjusted R-Square (0.812) suggest a relatively good fit of the model. These results highlight the strong relationship between the number of actual trips undertaken by divers and actual catch levels. In fact when plotting catch per trip for the period of estimation between 1998 and 2003 we observe a stable CPUE index. This phenomenon is widely observed for abalone and is referred to as ‘hyperstability’ (Annala et al. 2001). As previously noted, divers move among patches to maximise profits thereby maintaining a stable CPUE index where serial depletion masks true depletion rates. The results of the simple regression are not further used but serve to confirm the stable production function over the estimation period and harvestable areas. We can then translate the redistribution of trips predicted by the nested logit model due to closure directly into the predicted redistribution of catch and add it to the biological model by adopting the constant TACC level of 90t in 2003. The column “Change Trips (%)” in Table A2 in the Appendix is taken from the last column in Table 2, while the column “Trips (%)” shows the absolute number of trips due to closure of areas 4, 5, 9 and 16 in percentage terms. We can then calculate the absolute level of catch in tonnes that redistributes to the remainder of the areas as well as the decrease in overall catch due to closing the selected network of areas, as shown by the column “Catch (t)”.¹⁹

Initial bioeconomic simulations of closing areas 4, 5, 9 and 16 show the impact of closure derived from the nested logit model when assuming the predicted redistribution of catch (as shown by Table A2) constant for each year. Economic predictions focus on ‘first round’ impacts only, i.e., the nested logit model uses information on historic catch and weather data to predict a one-off round of redistributed effort in response to closing areas 4, 5, 9 and 16. This puts increased pressure on the remainder of the statistical areas within the QMA, especially in areas contiguous to the closed areas, and if assumed constant for more than a few years, leads to the collapse of stocks in areas 7, 8, 10, 11, 13 and 15 where exploitation rates range from 17 to 62%. The remainder of the areas experience relatively modest exploitation rates between 8 and 11%. To circumvent this problem we assume that after the first year of closure, effort will change in response to rising or declining legal biomass levels in the individual areas. We can anchor the model at the identified overall exploitation rate of 14.8% after the first year in each of the areas, i.e., we make the assumption of symmetrically shared

¹⁹ Table 2 illustrates the baseline prediction of the economic model in terms of the number of trips undertaken by divers. 21.3% of these ‘trips’ are assigned to NOTGO, i.e., divers decide not to go diving 21.3% of the choice occasions based on expected catch and weather conditions. It follows that when adopting the constant TACC level of 90t in 2003, then 90t really represents 78.7% of the total potential catch, the remainder of which has ‘not been harvested’ (synonymous to ‘NOTGO’). Total potential catch is hence 114t. The assumption of a potential catch above the TACC is a theoretical artefact, but is necessary to establish a link between the economic and the biological model.

Table 7 Closure of areas 4, 5, 9 and 16: bioeconomic model

	2003	2005	2007	2009	2010	2012	2113
C (t)	87	88	89	89	90	90	93
B _{Legal} (t)	680	708	731	753	774	794	987
B_{Legal fishable} (t)	588	595	599	603	606	609	626
B _{Spawn} (t)	1,030	1,060	1,088	1,115	1,140	1,164	1,377
R	853,193	865,911	876,143	884,832	892,305	898,725	937,202
TEgg × 10 ⁺¹³	2.503	2.635	2.753	2.863	2.968	3.068	4.087
TN (millions)	4.525	4.607	4.688	4.765	4.836	4.902	5.406
U (%)	14.8	14.8	14.8	14.8	14.8	14.8	14.8

information in PAU 5B (supported by personal experience with paua divers) where divers have perfect information about local abundance levels and respond to changes in area-specific biomass levels. The model is truly bioeconomic in the sense that it incorporates the initial disproportional shift of harvest to other areas predicted by the economic model in the first year ($U = 14.8\%$). Most importantly, in the years thereafter it captures the increase in overall pressure on legal biomass in the remainder of the areas. Table 7 shows overall PAU 5B biomass levels predicted by our bioeconomic simulations.

Table 7 has to be understood in the context of Tables 5 and 6. Catch as an indicator of profitability²⁰ highlights the undesirability of closed areas from a commercial fishing point of view, i.e., catch in Table 5 increases up to 106t in the long term while it remains under 94t for both closure scenarios. We now ask are there any conceivable circumstances under which a marine reserve could recommend itself in this fishery?

5.4 Spillover Gradients

The larval stages are the least understood stages of abalone recruitment and there is conflicting evidence with respect to the extent of larval dispersal. Some authors have argued for the local retention of larvae by observing demersal rather than pelagic dispersal of larvae (Prince et al. 1987; McShane and Smith 1988; McShane et al. 1988) but the majority of studies demonstrate wide larval dispersal of more than hundreds of metres (Sasaki et al. 1995; Shepherd and Breen 1992; Shepherd et al. 1992; Tegner 1992; Wells and Keesing 1990). Tissot (1992) points out that water movement is an important factor influencing the dispersal of abalone larvae. However, little is known about water current movements around Stewart Island. The National Institute of Water and Atmospheric Research (NIWA) reports a West–East flow, which reverses regularly.

We can then test the benefits of spillover for areas adjacent to the imposed no-take areas by introducing spillover gradients into the recruitment function. Despite studies demonstrating wide larval dispersal there is no literature or data from other fisheries (to the knowledge of the authors) to suggest a realistic larval spillover gradient in terms of recruit settlement. Larvae are very hard to locate during field samples as they are very patchy. The failure to detect recruits as well as the lack of field observations of larval movement make it hard to identify to what extent recruits on a parent reef are produced locally and to what extent they

²⁰ The use of catch as an indicator of profitability is based on the observed stable CPUE index for the period of estimation (as discussed at the beginning of this section) and fixed geographical distances to statistical areas.

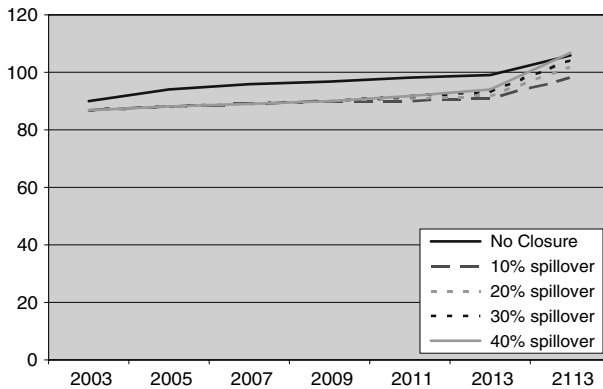


Fig. 2 Catch rates for spillover between 10 and 40%

have immigrated from other reefs (McShane et al. 1988). Bertness et al. (1992) investigate the variation in larval supply, settlement rates and settler mortality in two acorn barnacle populations, a species with characteristics similar to paua. Their findings suggest the co-existence of a localized stock–recruitment relationship with the dispersal of larvae to adjacent reefs, which is determined by a range of factors, such as adult density, suitable habitat, water movement, etc. Similarly, Minchinton and Scheibling (1991) show that 79% of the variation in barnacle recruit-density could be explained by larval availability in intertidal zones, while local hydrodynamics influence the distribution of remaining larvae that do not settle at the natal reef. However, none of these or other similar studies provide an estimate of possible spillover magnitudes. To account for the lack of knowledge of spillover gradients, we limit the effect of spillover to areas directly adjacent to closed areas. We model spillover gradients in the region of 10 to 40% where we ignore larval dispersal between fished statistical areas, i.e., we argue for density dependent dispersal from only the closed areas into statistical areas directly adjacent to them in both directions. Within the closed areas, we find legal biomass levels to rebound by at least 50% for all of the closed statistical areas within 10 years. Initial simulations under the assumption of density-dependent dispersal of larvae into statistical areas directly adjacent to closed areas show that after only 10 years (2013) legal biomass levels rise quickly, for example, for an assumed spillover gradient of 40% legal biomass levels increase by up to 13.6% in Area 10.

Table 8 illustrates catch and legal biomass levels predicted by bioeconomic simulations for the spillover gradients of 20 and 40% from the closed areas 4, 5, 9 and 16 compared to a no closure scenario with a fixed exploitation rate (Table 5). While legal biomass levels in the spillover scenarios never exceed biomass levels achieved by no closure, levels of catch highlight an interesting observation. In the long term, i.e., after 100 years, catch rates in all spillover scenarios approach the catch rate under no closure. Figure 2 shows that the effect is less marked for lower spillover gradients, e.g., with 20% spillover catch increases up to 102t in 2113 compared to 106t for no closure, while with 40% spillover catch (107t in 2113) actually exceeds the no closure scenario by a tonne. The results shown in Table 8 and Fig. 2 make the point that marine reserves might only be really effective in the long term. Short-term comparisons of catch and legal biomass will always lead to the conclusion that optimum management in terms of identifying a sustainable yield will be the preferred management tool, but in the very long term the losses of closed areas can be outweighed by assuming density-dependent spillover into adjacent areas.

Table 8 Spillover—comparison of catch and legal biomass levels

	2003	2005	2007	2009	2011	2013	2113
<i>No Closure (Table 5).</i>							
C (t)	90	94	96	97	98	99	106
BLegal (t)	661	686	701	712	721	730	782
<i>Closure areas 4, 5, 9 and 16 spillover 20%.</i>							
C (t)	87	88	89	90	91	92	102
BLegal fishable (t)	588	595	600	607	615	622	686
<i>Closure areas 4, 5, 9 and 16 spillover 40%.</i>							
C (t)	87	88	89	90	92	94	107
BLegal fishable (t)	588	595	600	611	623	634	720

5.5 Stochastic Recruitment

At face value the above results are in accordance with what bioeconomic models predict: marine reserves represent a strategy that comes only second-best to the perfect management scenario of fishing at a sustainable yield (Holland and Brazee 1996; Hannesson 1998). Even when assuming spillover gradients, the benefits are only apparent in the very long term. However, the collapse of the Californian abalone fishery in the late 1990s acts as a reminder of the complexities involved in managing sedentary species. That collapse has, amongst other things, been attributed to practising ‘pseudo-sustainable’ fishery management where fishing yields remained constant over a long time while serial depletion led to the successive failure of small-scale recruitment and localised depletion. Most stock assessment models, including our model, use a very simple deterministic equation to capture the recruitment relationship but empirical studies indicate highly variable recruitment dependent on a range of factors, both exogenous (e.g., due to environmental conditions) and endogenous (i.e., harvest-induced) shocks. For example, Sainsbury (1982) showed that the reproductive cycle for paua at Peraki Bay (New Zealand) is very irregular, where paua spawned for two successive years but subsequently failed to spawn for two years. Spawning cycles of paua were also shown to be variable at Banks Peninsula and Kaikoura (Wilson and Schiel 1995). In 1968 Poore (1973) found that the two paua species *H. iris* and *H. australis* spawned late summer–autumn, and *H. australis* had a second spawning during spring, however, neither species spawned during the following year. The reproductive cycle of paua remains poorly understood, and without incorporating the uncertainty inherent to wild populations (Cohen 1966), and the possible mitigating effects of marine reserves in the face of unexpected shocks and management failures, deterministic models will likely understate the value of no-take areas (Gerber et al. 2003).

The thrust of our argument focuses on the effects of biomass levels when accounting for stochastic shocks to recruitment. Ideally, we would simulate the irregularity of the reproductive cycle for paua in succeeding years but with little scientific knowledge about the long-term recruitment behaviour of paua stocks we consider it more useful to retain the assumption of yearly recruitment. The stochastic shock then relates to the varying levels of recruits produced each year. In order to reflect recruitment variability in terms of number of recruits

produced each year, [Breen et al. \(2000\)](#) include a stochastic component in the Beverton–Holt recruitment function as follows.

$$R_{t+1} = B_t^{spawn} / (\alpha + \beta B_t^{spawn}) \exp(Rdev_t - 0.5\sigma_R^2) \quad (14)$$

$Rdev_t$ is the recruitment residual for year t , i.e., it measures the difference in number of recruits predicted by the deterministic Beverton–Holt recruitment function and the actual number of recruits produced, given the stochastic nature of the marine environment. We cannot observe the actual number of recruits and thus have to generate a random sample reflecting the possible recruitment residual in any given year. We generate 30 streams of random recruitment residuals $Rdev_t$ according to a normal distribution with zero mean and $\sigma_R = 0.6$ ([Breen et al. 2000](#)). Every stream contains more than 100 values, each of which is fed into the recruitment function to predict yearly biomass levels as far as 100 years ahead in time.

Initial trials with a fixed exploitation rate (implying the Ministry of Fisheries responds to yearly fluctuations in biomass levels due to stochastic recruitment by adjusting the TACC) show that fishable legal biomass levels rise and drop accordingly (to anywhere as low as 100t and as high as 3,000t, depending on the assumed standard deviation of the recruitment residual) without ever leading to a population collapse. One possible concern then could be the Allee effect, also called depensation, which describes the observation that at low density and few aggregations of sessile broadcast spawners such as paua, reproductive success is reduced due to low fertilisation of gametes (mixing of eggs and sperm) because individuals are too far apart ([Allee 1931](#)). This implies there could be a biomass threshold, below which the population collapses even if fishing ceases altogether (critical depensation). We test for depensation and find that depending on the assumed depensatory parameter, paua populations are vulnerable to stock collapse at spawning biomass levels as low as 0.2–10t (see [Schneider 2006](#) for a more detailed discussion). This threshold is too low to have any real effect on our bioeconomic simulations under the assumption of responsive TACC adjustments. It is then interesting to see what happens when such adjustments are not made, possibly due to incorrect biomass estimates or political pressure to maintain the current TACC, i.e., we assume a constant catch of 90t as in [Table 4](#). With legal fishable biomass, B_{Legal} , as the variable of interest simulation results show that in some instances the PAU 5B stock is predicted to collapse when allowing for stochastic recruitment. [Figures 3 and 4](#) show frequency distributions for the legal biomass levels for the years 2013 and 2113, respectively. We picked these years since we are interested in the long-term viability of fisheries management, rather than short-term benefits. [Figure 3](#) shows the legal biomass level in 2013 is most frequently predicted to lie in the range of 700t to 800t (with a mean and median of 742t and 735t, respectively, and a standard deviation of 198t). This is comparable to the deterministic estimate of 774t in [Table 5](#). [Figure 4](#), however, highlights the extreme variability of predicted legal biomass levels beyond a 10-year timeframe (but within 100 years). At one extreme, the PAU 5B stock is predicted to collapse 12 out of 30 times (40%) after 15 to 100 years. At the other extreme, legal biomass levels are predicted to rise to over 2,000t for 6 out of 30 times (20%). As expected, the standard deviation is very high (936t) and there is a wide range of possible biomass estimates (2,906t) due to the possibility of collapse, where biomass levels are treated as having the value zero.

The conclusions drawn hinge to a large extent on the stochastic component added to the recruitment function. We can test how sensitive estimated biomass levels are to changes in the assumed standard deviation of 0.2 at one extreme and of 1 at the other. For a lower standard deviation of $\sigma_R = 0.2$ and a sample of 10 streams, biomass estimates show very little variation and no stock collapses (e.g., biomass levels have a standard deviation of

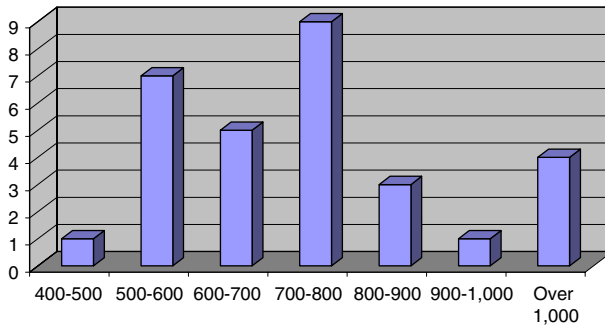


Fig. 3 Frequency distribution for BLegal in 2013

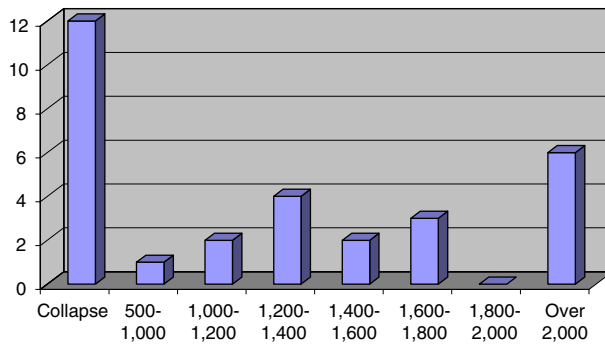


Fig. 4 Frequency distribution for BLegal in 2113

only 162t in 2113). However, when assuming a standard deviation of $\sigma_R = 1$ recruitment residuals are much more likely to lead to a stock collapse in the long term, i.e., 70% of the time our simulations predict a stock collapse in the long-term. [Lauck et al. \(1998\)](#) obtain similarly striking results analytically when assuming uncontrollable harvest represented by a probability distribution. They find that even when assuming a moderate coefficient of variation, the chance of successful management (defined as the probability that the stock remains above 60% of its carrying capacity over some time interval) drops rapidly from 1 once the harvestable area size increases to above 30% of the total area. When assuming a large coefficient of variation, a fishable area even as small as 5% of the total area implies a chance of success below 1.

We conduct further trials by focusing on α , which is crucial to estimating the population growth. We calibrated the model to match legal biomass estimates by [Breen et al. \(2000\)](#) by setting $\alpha = 655$. The assumed value for α underpins all simulated legal biomass and catch levels and sensitivity trials show that predicted biomass levels differ to some extent when varying α between 500 and 750 in steps of 50. Picking two streams from the 30 generated above (where one stream implies a large ‘positive’ random shock with legal biomass levels growing to 2,905t in 2113 under the assumption of constant catch, while the other stream represents a large ‘negative’ shock where the stock is predicted to collapse after 14 years), we find that for $\alpha = 500$ legal biomass levels are predicted to be nearly twice as high on average (BLegal=2,136t) than for $\alpha = 750$ (BLegal=1,322t) for the first stream, while the prediction of stock collapse applies to nearly all cases apart from $\alpha = 500$ for the latter stream. Overall, qualitative predictions are fairly robust in that the first stream predicts positive legal biomass

rates for all chosen α , while the other stream predicts a long-term stock collapse for all chosen α (except for $\alpha = 500$), however, quantitative predictions of legal biomass differ according to the level of α .

Our sensitivity analysis shows that, when taking account of inherently variable environmental conditions, which will affect recruitment, predictions about stock status and the impact of different harvest levels become much more difficult. The PAU 5B stock is predicted to collapse 12 out of 30 times (40%) after 15–100 years when assuming a standard deviation of 0.6 for the generated streams of recruitment residuals. One could argue that it is more realistic to assume divers respond to changes in biomass and the TACC would likely be reduced to halt any conceivable stock collapse. While this might be true, our analysis has highlighted the fact that the accurate assessment of a stock very much depends on the underlying assumptions of the recruitment function. Scientists are well aware of the large uncertainties when recommending quotas, but fisheries managers are reluctant to rely on anything other than scientific best point estimates. It is obviously true that TACCs could be set very conservatively to minimize risk, however, the uncertainties and political biases associated with setting quotas and determining the actual fishing mortality imply that the fishery would probably remain vulnerable to overexploitation unless quotas were set far below best point estimates (Lauck et al. 1998). Personal communications with scientists at the Ministry of Fisheries confirm the fact that in the face of recruitment variability a TACC should ideally be set extremely conservatively to minimise risk. The collapse of the Californian abalone fishery in the late 1990s acts as a reminder of the complexities involved in managing sedentary species.

Conrad (1999) compared the effects of a marine reserve on the variation in biomass assuming uncorrelated and perfectly correlated stochastic net growth between the fished and unfished areas in the presence of migration with the main result that, with perfect correlation, migration was not able to reduce the variation in biomass to the degree observed when stochastic net growth in each of the areas was uncorrelated. We run a similar analysis by comparing the stochastic components for each of the randomly generated streams, and varying them between the fished and closed areas. However, the reduction in the variation in biomass as predicted by Conrad (1999) remains unverified (See Schneider 2006 for a more detailed discussion). Grafton et al. (2004) incorporated stochasticity into their dynamic sink-source model to show that reserves have value under uncertainty not only because they increase population persistence and reduce the variance of populations and harvests, but primarily because they increase resilience, or the speed it takes a population to return to a former state following a negative shock. Again, a comparison of results between perfectly correlated and uncorrelated stochastic recruitment in our case shows no marked improvements in resilience. The absence of reduced biomass variation and the weak resilience effect can most likely be attributed to the relatively small area that is assumed closed in comparison to the fished areas. Statistical areas 4, 5, 9 and 16 make up 1.54% of the whole of PAU 5B. Compared to analytical studies, which look at closures in the range of 10–50%, our area under consideration is likely to be too small to show any marked impacts on biomass variability and resilience. In addition, unlike Conrad (1999), Grafton et al. (2004) and other authors, we incorporate the redistribution of effort in the areas adjacent to the closed areas, which dampen some of the benefits predicted by analytical models.

Given our analysis, marine reserves, or a network of no-take areas, would raise the likelihood that the population contained within the closed areas will not be eliminated if the remainder of the fished areas collapse due to harvest-induced shocks that are not reflected in stock assessments. Table 9 provides the results of a hypothetical scenario. We assume the statistical areas 4, 5, 9 and 16 are closed to fishing while the remainder of the areas experience a massive harvest-induced shock to recruitment so that paua populations crash. This scenario

Table 9 Time taken to rebuild legal biomass after collapse

	B _{Legal} = 200t	B _{Legal} = 400t	B _{Legal} = 600t
No Spillover	56 years	62 years	66 years
10% Spillover	21 years	26 years	30 years
20% Spillover	16 years	20 years	23 years
30% Spillover	14 years	17 years	20 years
40% Spillover	12 years	15 years	17 years

is not too far fetched given the possibility of serial depletion and the problems associated with accurate stock assessments of sedentary populations that occur in large aggregations. Assuming simulated legal biomass levels in areas 4 (9t), 5 (32t), 9 (15t) and 16 (20t) as of 2002 we can estimate the number of years it would take for legal biomass levels in the fished areas to rebuild, i.e., no harvesting occurs during the rebuild phase. Table 9 shows that assuming no spillover, it would take 66 years for fishable legal biomass levels to rebuild to roughly 600t, but only 17 years assuming a spillover gradient of 40%. Even for a modest spillover of 10% it would take only half of the time for stocks to rebuild compared to the assumption of no spillover. The introduction of no-take areas as insurance to increase the likelihood of population persistence would be in the spirit of the precautionary approach, which is requested in a number of international instruments of importance to fisheries; for example the General Principles and Article 6.5 of the 1995 FAO International Code of Conduct for Responsible Fisheries prescribes a precautionary approach to all fisheries, in all aquatic systems, and regardless of their jurisdictional nature, recognizing that most problems affecting fisheries result from insufficiency of precaution in management regimes when faced with the high levels of uncertainty encountered in fisheries. This discussion can be expanded to include scenarios where biomass levels rebuild at different more realistic levels, but this is beyond the scope of this paper and is left for future research.

6 Conclusion

Advocates of marine reserves have often focussed exclusively on biology and ignored economics. Sustainability of human societies requires an understanding of the feedback between ecological and economic systems. In this paper we have applied bioeconomic modelling to a New Zealand paua fishery where the threat from over-fishing is ever present. Our model of diver behaviour indicates the importance of taking into account rational economic responses to the imposition of spatial closures. The economic model predicts a redistribution of effort to areas with high relative catch, which is in accordance with Smith and Wilen's (2003) findings, but the results are not as pronounced due to the absence of de facto marine reserves. Smith and Wilen (2003) compare a number of scenarios including the prediction of an increase in steady-state harvest by 40% in response to closure based on a model excluding economic behaviour, which stands in contrast to a drop in steady-state harvest by 10% once the economic component is included. We find that, in the absence of spillover, bioeconomic deterministic simulations of closing a selected network of areas always leads to lower catch levels, whether the economic model is included or not. Only when introducing spillover gradients into areas directly adjacent to the closed areas can catch rates be seen to converge

to those without closure in the long term. The management regime differs to Smith and Wilen's (2003) case study of the northern California red sea urchin fishery where regulatory restrictions include closed seasons, minimum size limits and a limited entry program. The authors report a baseline participation rate of only 14% of available open days, while our dataset implies a participation rate of approximately 75%. This discrepancy can be explained by the fact that paua is managed under New Zealand's ITQ system (in addition to restrictions on gear use and a minimum size limit), where quota holders/leasers finely tune their quota holdings to their diving activity.²¹ In addition, little intra-annual variation in the beach price of paua is likely to have a smoothing effect on diving activity throughout the year unlike for the red sea urchin fishery, where overnight shipping to Japan yields a price that is highly dependent on the Tokyo Central Wholesale Market.

On first analysis thus it would seem that our deterministic bioeconomic model indicates very little is to be gained in the management of the Stewart Island abalone fishery by the introduction of a network of no-take areas to supplement the current quota management system. However, we extend Smith and Wilen's (2003) work by taking into account the uncertainty of the marine environment and how that can contribute to variations in recruitment to the fishery. We find that depending on the assumed standard deviation of the recruitment residual the PAU 5B stock is predicted to collapse 0, 40 and 70% times for low, moderate and high assumed standard deviations, respectively, within a timeframe of 100 years. These results are comparable to Lauck et al.'s (1998) predicted diminishing chances of successful harvest management as the coefficient of variation is increased. We conclude that once the inherent variability of the environment is recognised, it can be demonstrated that a marine reserve could guard against surprises arising from our lack of certain knowledge and could do so at minimal economic cost. Grafton et al. (2005) note that, without incorporating uncertainty in wild populations and the possible mitigating effects of marine reserves in the face of unexpected shocks and management failures, deterministic models will likely understate the value of no-take areas. Our results support this statement. In the short and medium term deterministic bioeconomic simulations show that marine reserves are costly in terms of lower catch rates. However, stochastic bioeconomic simulations imply that they can offer an insurance policy or hedge against the combined effects of irreducible uncertainty and management limitations.

The implications for the paua fishery PAU 5B around Stewart Island are far-reaching. A network of no-take areas as an additional management tool to New Zealand's quota system could ensure the long-term sustainability of the paua fishery given its biological characteristics, the uncertainties surrounding quota recommendations in the face of serial depletion, the lack of knowledge concerning recruitment and parameter uncertainty. More detailed information would improve the accuracy of predictions made by the bioeconomic model. For example, additional information on expected prices, time elapsed since the last day fished and individual diver characteristics could be included in the discrete choice analysis. Similarly, biological information on natural mortality and the relationship between standing stock and recruitment of paua are still poorly understood and require further research to provide more accurate stock assessments. Furthermore, economic predictions focus on 'first round' impacts only and increased pressure on the fished areas can lead to unsustainable exploitation

²¹ Personal communication with staff at the Ministry of Fisheries and limited data access to the quota history of PAU 5B between 2001 and 2006 revealed that PAU 5B quota trading activity was very low since the introduction of the ITQ system in 1997, i.e., no major shifts via exit from and entry to PAU 5B occurred. Instead, market activity has evolved around the leasing of ACE (annual catch entitlements) to small scale fishermen and medium sized companies. The data show that the maximum number of ACE holders active throughout the year between 1998 and 2003 remained fairly stable at around 22–30.

if assumed constant for more than a few years. We circumvent this problem by assuming a constant exploitation rate but this is problematic due to imperfect information and administrative lags. Further research addressing such issues would be a valuable addition to the assessment of spatial management tools.

Acknowledgements The authors thank two anonymous referees for helpful comments and suggestions that improved the quality of the paper.

Appendix

Table A1 Sensitivity to changes in M

BC: $L_\infty = 150, K = 0.252$	$(h = 0.4, R_{\text{coff}} = 13.971, R_0 = 1, 168, 230)$		
M	0.108	0.12	0.132
Virgin legal biomass (t)	3,004	2,496	2,099
Virgin spawning biomass (t)	3,510	2,983	2,566
Total egg production	1.476×10^{14}	1.190×10^{14}	9.730×10^{13}
Total number (<i>million</i>)	10.229	9.157	8.276
α	1,127	957	824
β	5.35×10^{-7}	5.35×10^{-7}	5.35×10^{-7}
LO: $L_\infty = 140, K = 0.343$	$(h = 0.4, R_{\text{coff}} = 14.37, R_0 = 1, 741, 052)$		
M	0.108	0.12	0.132
Virgin legal biomass (t)	4,171	3,465	2,914
Virgin spawning biomass (t)	4,987	4,250	3,669
Total egg production	1.864×10^{14}	1.497×10^{14}	1.221×10^{14}
Total number (<i>million</i>)	15.245	13.647	12.334
α	1,074	915	790
β	3.59×10^{-7}	3.59×10^{-7}	3.59×10^{-7}
HI: $L_\infty = 160, K = 0.198$	$(h = 0.4, R_{\text{coff}} = 13.67, R_0 = 864, 581)$		
M	0.108	0.12	0.132
Virgin legal biomass (t)	2,327	1,932	1,623
Virgin spawning biomass (t)	2,691	2,282	1,958
Total egg production	1.243×10^{14}	1.005×10^{14}	8.233×10^{13}
Total number (<i>million</i>)	7.570	6.777	6.125
α	1,167	990	849
β	7.23×10^{-7}	7.23×10^{-7}	7.23×10^{-7}

Table A2 Redistribution of catch predicted by the economic model

	Areas 4, 5, 9 and 16 closed		
	Change trips (%)	Trips (%)	Catch (t)
NOTGO (no catch)	3.32	24.62	28.15
B1	0.94	5.52	6.32
B2	0.89	5.15	5.89

Table A2 continued

	Areas 4, 5, 9 and 16 closed		
	Change trips (%)	Trips (%)	Catch (t)
B3	0.73	4.18	4.78
B4	-2.34	0	0
B5	-3.76	0	0
B6	0.86	5.15	5.89
B7	1.07	6.45	7.37
B8	1.19	6.89	7.88
B9	-5.70	0	0
B10	1.24	7.24	8.28
B11	1.32	7.64	8.74
B12	1.35	7.87	9.00
B13	1.11	6.64	7.59
B14	1.13	6.65	7.60
B15	1.04	6.00	6.86
B16	-4.39	0	0
<i>Potential</i>	<i>100</i>	<i>100</i>	<i>114.36</i>
Actual	0.00	75.38	86.20

References

- Alcala AC, Russ GR (1990) A direct test of the effects of protective management on abundance and yield of tropical marine resources. *Journal du Conseil International pour l'Exploration de la Mer* 46:40–47
- Allee WC (1931) Animal aggregations. University of Chicago Press, Chicago
- Allison GW, Lubchenco J, Carr MH (1998) Marine reserves are necessary but not sufficient for marine conservation. *Ecol Appl* 8(1):Supplement S79–S92
- Anderson LG (2002) A bioeconomic analysis of marine reserves. *Nat Resour Model* 15(3):311–334
- Annala JH, Sullivan KJ, O'Brien CJ, Smith NWMcL (2001) Report from the Fishery Assessment Plenary, May 2001: Stock Assessments and Yield Estimates. Ministry of Fisheries, Wellington, New Zealand
- Annala JH, Sullivan KJ, Mace PM, Smith NWMcL, Griffiths PR, Livingston ME, Harley SJ, Key JM, Connell AM (2005) Report from the Fishery Assessment Plenary, May 2005: stock assessments and yield estimates. Ministry of Fisheries, Wellington, New Zealand
- Bertness MD, Gaines SD, Stephens EG, Yund PO (1992) Components of recruitment in populations of the acorn barnacle *Semibalanus balanoides* (Linnaeus). *J Exp Mar Bio Ecol* 156:199–215
- Beverton RJH, Holt SJ (1957) On the dynamics of exploited fish populations. *Fish Invest (Ser II)* 19:1–533
- Bockstael NE, Opaluch JJ (1983) Discrete modelling of supply response under uncertainty: the case of the fishery. *J Environ Econ Management* 10:125–137
- Breen PA, Andrew NL, Kendrick TH (2000) Stock assessment of paua (*Haliotis iris*) in PAU 5B and PAU 5D using a new length-based model. New Zealand Fisheries Assessment Report 2000/33
- Cohen D (1966) Optimizing reproduction in a randomly varying environment. *J Theoret Biol* 12:119–129
- Conrad JM (1999) The bioeconomics of marine sanctuaries. *J Bioecon* 1:205–217
- Christie P, White A, Deguit E (2002) Starting point or solution? Community-based marine protected areas in the Philippines. *J Environ Manage* 66:441–454
- DeMartini EE (1993) Modeling the potential of fishery reserves for managing Pacific coral reef fishes. *Fish Bull* 91:414–427
- Gerber LR, Botsford LW, Hastings A, Possingham HP, Gaines SD, Palumbi SR, Andelman S (2003) Population models for marine reserve design: a retrospective and prospective synthesis. *Ecol Appl* 13(1):Supplement S47–S64

- Grafton RQ, Kompas T, Ha PV (2005) The economic payoffs from marine reserves: resource rents in a stochastic environment. Asia Pacific School of Economics and Government Working Papers, the Australian National University
- Guenette S, Lauck T, Clark C (1998) Marine reserves: from Beverton and Holt to the present. *Rev Fish Biol Fish* 8:251–272
- Halpern BS (2003) The impact of marine reserves: do reserves work and does reserve size matter? *Ecol Appl* 13(1): Supplements S117–S137
- Hannesson R (1998) Marine reserves: what would they accomplish?. *Mar Resour Econ* 13:159–170
- Hausman J, McFadden D (1984) Specification tests for the multinomial logit model. *Econometrica* 52:1219–1240
- Hensher DA, Greene WH (2000) Specification and estimation of the nested logit model: alternative normalizations. Working paper, Department of Economics, Stern School of Business, New York University
- Hicks RL, Kirkley J, Strand IE (2004) Potential short-run welfare losses from essential fish habitat designations for the surfclam and ocean quahog fisheries. *Mar Resour Econ* 19(1):113–129
- Hilborn R (2002) Marine reserves and fisheries management. *Science* 295:1233–1235
- Holland DS (2000) A bioeconomic model of marine sanctuaries on Georges Bank. *Can J Fish Aquat Sci* 57:1307–1319
- Holland DS, Brazee RJ (1996) Marine reserves for fisheries management. *Mar Resour Econ* 11:157–171
- Lauck T (1996) Uncertainty in fisheries management. In: Gordon DV, Munro GR (eds) *Fisheries and uncertainty: a precautionary approach to resource management*. University of Calgary Press, Chapter 5
- Lauck T, Clark CW, Mangel M, Munro GR (1998) Implementing the precautionary principle in fisheries management through marine reserves. *Ecol Appl* 8(1):S72–S78
- Minchinton TE, Scheibling RE (1991) The influence of larval supply and settlement on the population structure of barnacles. *Ecology* 72(5):1867–1879
- McFadden D (1973) Conditional logit analysis of qualitative choice behavior. In: Zarembka P (ed) *Frontiers in econometrics*. Academic Press, New York, pp 105–142
- McShane PE (1995) Estimating the abundance of abalone: the importance of patch size. *N Z J Mar Freshw Res* 46:657–662
- McShane PE, Smith MG (1988) Measuring abundance of juvenile abalone *Haliotis ruber* Leach (Gastropoda Haliotidae); comparison of a novel method with two other methods. *Aust J Mar Freshw Res* 39:331–336
- McShane PE, Black KP, Smith MG (1988) Recruitment processes in *Haliotis rubra* (Mollusca: Gastropoda) and regional hydrodynamics in southeastern Australia imply localized dispersal of larvae. *J Exp Mar Biol Ecol* 124:175–203
- McShane PE, Mercer SF, Naylor JR (1994) Spatial variation and commercial fishing of New Zealand abalone (*H. iris* and *H. australis*). *N Z J Mar Freshw Res* 28:345–355
- Mosquera I, Cote IM, Jennings S, Reynolds JD (2000) Conservation benefits of marine reserves for fish populations. *Anim Conserv* 4:321–332
- Myers RA (2001) Stock and recruitment: generalizations about maximum reproductive rate, density dependence, and variability using meta-analytic approaches. *ICES* 58:937–951
- Pezzey JCV, Roberts CM, Urdal BT (2000) A simple bioeconomic model of a marine reserve. *Ecol Econ* 33:77–91
- Polacheck T (1990) Year around closed areas as a management tool. *Nat Resour Model* 4(3):327–354
- Poore GCB (1973) Ecology of New Zealand abalones, *Haliotis* species (Mollusca: Gastropoda). *N Z J Mar Freshw Res* 1(1 & 2):67–84
- Prince JD, Sellers TL, Ford WB, Talbot SR (1987) Experimental evidence for limited dispersal of haliotid larvae (genus *Haliotis*, Mollusca: Gastropoda). *J Exp Mar Biol Ecol* 106:243–263
- Roberts CM, Polunin NVC (1991) Are marine reserves effective in management of reef fisheries?. *Rev Fish Biol Fish* 1:65–91
- Roberts CM, Polunin NVC (1993) Marine reserves: simple solutions to managing complex fisheries?. *Ambio* 22(6):363–368
- Roberts CM, Bohnsack JA, Gell F, Hawkins JP, Goodridge R (2001) Effects of marine reserves on adjacent fisheries. *Science* 294:1920–1923
- Rowley RJ (1992) Impacts of marine reserves on fisheries: a report and review of the literature. Science & Research Series No. 51, Department of Conservation, Wellington, New Zealand
- Sainsbury KJ (1982) Population dynamics and fishery management of paua, *Haliotis iris*. I. Population structure, growth, reproduction and mortality. *N Z J Mar Freshw Res* 16:147–161
- Sanchirico JN (2005) Additivity properties in metapopulation models: implications for the assessment of marine reserves. *J Environ Econ Manage* 49(1):1–25
- Sanchirico JN, Wilen JE (1998) Marine reserves: is there a free lunch? Resource for the Future, Discussion Paper 99–09

- Sasaki R, Shepherd SA (1995) Larval dispersal and recruitment of *Haliotis discus hannai* and *Tegula* spp. on Miyagi coasts, Japan. *Mar Freshw Res* 46:519–529
- Schiel DR, Breen PA (1991) Population structure, ageing, and fishing mortality of the New Zealand abalone *Haliotis iris*. *Fish Bull* 89:681–691
- Schneider V (2006) A bioeconomic analysis of marine reserves for paua (abalone) management at Stewart Island, New Zealand. PhD thesis, University of Otago, New Zealand, May 2006
- SeaFIC (2003) Exports of seafood produce for 12 months ending December 2002. NZ Seafood Industry Council (SeaFIC), www.seafood.co.nz, accessed August 2003
- Shepherd SA, Breen PA (1992) Mortality in abalone: its estimation, variability and causes. In: Shepherd SA, Tegner MJ, Guzman del Proo S (eds) *Abalone of the world: biology, fisheries, and culture*. Blackwell Scientific
- Shepherd SA, Brown LD (1993) What is an abalone stock: implications for the role of refugia in conservation. *Can J Fish Aquat Sci* 50:2001–2009
- Shepherd SA, Lowe D, Partington D (1992) Studies on southern Australian abalone (genus *Haliotis*) XIII: larval dispersal and recruitment. *J Exp Mar Biol* 164:247–260
- Smith MD, Wilen JE (2003) Economic impacts of marine reserves: the importance of spatial behavior. *J Environ Econ Management* 46(2):183–206
- Smith MD, Zhang J, Coleman FC (2006) Effectiveness of marine reserves for large-scale fisheries management. *Can J Fish Aquat Sci* 63:153–164
- Tegner MJ (1992) Brood-stock transplants as an approach to abalone stock enhancement, In: Shepherd SA, Tegner MJ, Guzman del Proo S (eds) *Abalone of the world: biology, fisheries, and culture*. Blackwell Scientific
- Tissot B (1992) Water movement and the ecology and evolution of the *Haliotidae*. In: Shepherd SA, Tegner MJ, Guzman del Proo S (eds) *Abalone of the world: biology, fisheries, and culture*. Blackwell Scientific
- Tupper MH (2002) Marine reserves and fisheries management. *Science* 295(5558):1233–1235
- Ward TJ, Heinemann D, Evans N (2001) The role of marine reserves as fisheries management tools: a review of concepts, evidence and international experience. Bureau of Rural Sciences, Canberra, Australia
- Wells FE, Keesing JK (1990) Population characteristics of the abalone *Haliotis roei* on intertidal platforms in the Perth metropolitan area. *J Malacol Soc Aust* 11:65–71
- Wickstrom K (2002) Marine reserves and fisheries management. *Science* 295(5558):1233–1235
- Wilson NHF, Schiel DR (1995) Reproduction in two species of abalone (*Haliotis iris* and *H. australis*) in southern New Zealand. *Mar Freshw Res* 46:629–637