ISSN 0111-1760

University of Otago
Economics Discussion Papers
No. 0407

June 2004

"Simulating the effects of marine reserves as an additional management tool to paua harvest around Stewart Island: an economic analysis."

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[*] I would like to thank Robert Alexander for the useful feedback he gave me for this paper.

Abstract

Marine reserves have increasingly been recognised for their potential to address the pervasive problem of unsustainable harvest of fisheries worldwide. Abalone catch around Stewart Island has been in decline since the turn of the century despite being managed under NZ's quota system. A nested logit model is applied to spatially recorded catch and effort data between 1998 and 2003 to capture the two level decision making process of divers. On any given day, divers decide whether to go diving at all, and if yes, which of the 16 statistical areas around Stewart Island to visit. Weather conditions, patch-specific levels of catch per unit of effort and distance used as explanatory variables lead to a reasonably good fit, and the model can be used to simulate the imposition of a network of no-take areas as an additional management tool to NZ's quota system. Suitable areas are selected according to criteria that yield the 'least economic impact'. The results show that most effort redistributes to adjacent areas, which could potentially outweigh the benefits of no-take areas. This highlights the fact that incorporating fishermen behaviour is crucial in the marine reserve debate.

Keywords: management tool, marine reserve, nested logit model, network, simulation;

I. Introduction

In 2000 the Food and Agriculture Organization of the United Nations estimated 18% of the world's commercial fish stocks to be fully exploited and 47% to be overexploited (FAO 2000). Since World War II, traditional fishing control methods, such as gear restrictions, limited entry, taxes, etc., have been the approach preferred by fisheries managers because they are easy to understand and enforcement was believed to be sufficiently effective to control fishing mortality. More than half a century later, fisheries science has come to a crossroads. Despite elaborate and intensive management efforts, commercially valuable fish stocks are dwindling. For example, the recent northern cod disaster on the Grand Banks in Canada took place despite a high degree of harvest control by a government committed to moderate exploitation and employing world-class marine scientists (Hannesson 1998). Lauck *et al.* (1998) point to the inherent and persistent scientific uncertainty that surrounds the marine environment, which renders management controls of single fish species, bar multi-species systems, futile and leads to what is now commonly known as serial depletion, i.e. fishermen are successively moving down the marine food web by substituting lower trophic level fish species for higher level ones.

Marine reserves or so called no-take areas, which are closed to all forms of fishing, have increasingly been recognized for their bifold potential as ecosystem preservation and fisheries management tools (Roberts & Polunin 1991, Rowley 1994). Field studies from widespread international sites support predictions by biologists that marine reserves foster increased densities and larger average sizes of fish within reserves (e.g. Buxton & Smale 1989, Chapman & Kramer 1999, Edgar & Barrett 1999, etc.). Rowley (1994) points to the potential of increased fishery yields, either by emigration of large fish across the border of reserves into adjacent fishing patches or by the export of larvae, which may enhance recruitment. Some studies provide direct evidence of enhanced yields in adjacent fisheries (e.g. Alcala & Russ 1990, Roberts *et al*. 2001), but effort tends to concentrate on the borders of reserve boundaries reversing the beneficial effects of enhanced stocks (McClanahan & Mangi 2000). Murawski *et al*. (2000) demonstrate that the effectiveness of closures in temperate marine systems crucially depends on the co-existence of stringent, complementary management regulations limiting the amount of effort.

The paucity of direct evidence of the fishery yield benefits provided by no-take areas has prompted theoretical modelling studies, mainly in the ecological domain, which more than often neglect to incorporate economic behaviour (Wilen & Smith 2001). Some interdisciplinary studies originating in the biological literature do account for economic variables (Beverton & Holt 1957, Polacheck 1990, DeMartini 1993 and Pezzey, Roberts & Urdal 2000), but approach the problem mainly from a natural science angle. Recently, economic representations by Conrad (1999), Holland & Brazee (1996), Hannesson (1998) and Sanchirico & Wilen (1999) of simulating the benefits of reserves, given certain biological parameters, have added to the literature. Bockstael & Opaluch (1983) used discrete choice modelling for the first time to analyse and predict fishers' location choices. This approach is fairly novel to fishery economics and has been adopted by economists such as Holland (2000) and Wilen & Smith (2001) to predict the behavioural responses of fishermen to the imposition of no-take areas.

Given the poor management performance of paua around Stewart Island, NZ, within recent years, a nested logit model is applied to identify the determinants of diver behaviour. Paua has been managed under NZ's quota system since the mid 1980s, and while enforceable quotas do provide a valuable tool in limiting effort, the particular biological characteristics of paua suggest the need for additional spatial management measures. In absence of any biological information, we simulate the imposition of a network of no-take areas according to the criterion of 'least economic impact' in order to prove policy direction as to the effect of such an additional management tool on participation and effort distribution decisions. The outline of the remainder of this paper is as follows. Section II provides background information on the NZ Paua industry, specifically the Stewart Island fishery. Section III describes the data set, and section IV outlines the modelling approach used in deriving the results and simulations presented in sections V and VI respectively. Section VII concludes and adds some general comments on the viability of marine reserves as an additional management tool.

II. The NZ paua industry

The genus *Haliotis*, more commonly known by its North American name abalone, is characterised by a flattened, ear shaped shell containing more than 100 species worldwide, which are most common throughout the temperate and sub-tropical waters (Hahn 1989). The majority of these subtidal algal grazers, both in terms of abundance and diversity of species, can be found in Australia, western North America, New Zealand, Japan and the Pacific Islands (McShane 1991). In New Zealand abalone are more commonly known by their Maori name, paua. There are three species endemic to the shallow rocky coastlines of the North Island and South Island, as well as Stewart Island, Snares Island and the Chatham Islands: *Haliotis iris* (also know as the blackfoot paua), *Haliotis australis* (yellowfoot) and *Haliotis virginea* (whitefoot) (Sainsbury 1982). Paua occupies rocky intertidal and subtidal habitats to a depth of 10-15 meters (Wilson 1987).

The commercial abalone fishery began shortly after World War II and was initially developed to encourage employment of war veterans in the remote parts of New Zealand. Profits were primarily derived from the iridescent shells used in jewellery and souvenirs, while the meat was discarded due to lack of demand (Naysmith 2000). Throughout the mid 1960s the small domestic market slowly branched out to Southeast Asia, where canned abalone meat gained popularity. During the late 1960s and early 1970s the fishery underwent rapid growth in paua export demand and large commercial beds came under intensive fishing pressure (Murray 1982). In an attempt to restrain effort, export restrictions on shell and meat products were imposed in 1973, but entry into the fishery remained unregulated, and by 1977 the number of abalone permit holders had nearly doubled from 253 to 402 (Harrington 2000). Export restrictions were eased in 1980, 1986 and again in 1990 to allow for the export of shells and live abalone[1]. By the mid 1980s growing concern about the sustainability of paua resources led to the adoption of *Haliotis* into the quota management system (QMS). The Ministry of Fisheries determines a yearly nationwide total allowable catch (TAC), which is set according to the estimated maximum sustainable yield. Additional management tools include a minimum legal size of 125mm for blackfoot abalone and 80mm for yellowfoot abalone[2], as well as gear restrictions. Divers are banned from using SCUBA to collect paua, and recreational paua catch, which has increasingly been putting strain on the efficiency of the quota system, is limited to 10 paua per person per day.

The choice of paua to simulate the effects of no-take areas as an additional management tool is motivated by a number of factors. Abalone is a valuable resource worldwide, as well as nationally. The growth of international markets has led to intensified fishing pressure, and the wild fisheries of abalone have been showing symptoms of overfishing in terms of declining harvests worldwide over the past two decades[3] (Shepherd & Brown 1993). Over-harvesting, poor enforcement of fishery regulations and disease are seen as the major contributors, and some species, such as the Californian white abalone (*Haliotis sorenseni*), are on the brink of being listed as endangered (Tissot 2003). New Zealand is a key player in the international abalone market with the third largest production of over 1,000 metric tons per year as of 1999 (Australia and Japan claim the biggest share at over 5,000 and 2,000 metric tons per year,

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[1] Most of the abalone products are now supplied to Asian markets, where the firm flesh is a delicacy often served raw, in sushi or sashimi style.
[2] The whitefoot abalone is not commercially harvested.
[3] The decline of wild populations has led to the rapid development of aquaculture techniques. While many countries traditionally not associated with paua have taken up production, such as France, Iceland, etc., aquaculture development in NZ has been stagnant (Naysmith 2000).

respectively)[4]. 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 1986-87 fishing season (Annala *et al*. 2001). It is gathered by hand while free diving in shallow waters, usually not more than 6m off the coastline of New Zealand. Virtually all the commercial catch is made up of the blackfooted paua *H.iris*, with the majority of harvest being derived from the South Island (PAU 7), Stewart Island (PAU 5) and the Chatham Islands (PAU 4).

Paua 5, more specifically Paua 5B (PAU 5B) around Stewart Island, is the area of interest primarily because of its poor management performance in recent years. From 1995 onwards, PAU 5 was subdivided into the three QMAs of Fiordland (PAU 5A), Stewart Island (PAU 5B) and the Catlins/Otago (PAU 5D) for reporting purposes. These areas were further divided into 17, 16 and 11 statistical areas, respectively, in 1997. Annala *et al*. (2001) provide catch per unit of effort (CPUE) indices for the various paua management areas. While catch rates appear fairly stable in most of the areas (including PAU 5A and PAU 5D), CPUE in PAU 5B displays a steady decline between 1984 and 1999 as shown by Figure 1.

More recently, landings and quota levels shown by Figure 2 point to symptoms of unsustainable harvest as catch remains below the total allowable commercial catch (TACC) and has taken a marked decline since shortly before the turn of the century. Figure 2 shows that although NZ's market based quota system is hailed as one of the most effective regulative measures to promote sustainable catch, some species, such as paua in PAU 5B, show signs of overexploitation. Amidst problems of by-catch, high-grading, recovery rate and multi-species catch, by far the most significant problem the ITQ system faces is setting the correct level of TAC (Anderson 1995). Methods to estimate the TAC in the abalone fishery are based on

[4] In New Zealand, paua production is exported primarily to the western Pacific Rim where Singapore and Hong Kong together account for 80% of the total recipients. Nearly all of the paua is sold processed (canned) and was worth nearly NZ$60 million in 2002 (SeaFIC 2003).

yield- and egg-per-recruit models, which rely on the assumption that CPUE is some constant proportion of a homogeneously abundant stock (Breen 1992). However, Breen (1992) points out that CPUE is not a good index of paua abundance, i.e. stable catch rates do not necessarily indicate a stable biomass (Annala *et al*. 2001).

The reason for this is that paua are known to form large aggregations on reefs[5], possibly to enhance fertilization, shelter or feeding (McShane 1995). Divers tend to move amongst patches targeting large aggregations to maximise rent. Thus, fishing would result in a reduction of large patches while smaller patches would persist longer. McShane (1995) points out that a small reduction in the frequency of large aggregations would directly relate to a large decrease in relative abundance. Movement of adult paua occurs over a very small spatial scale so that they are considered sedentary (Annala *et al.* 2001). However, patches are loosely connected with each other so that local populations are aggregated into metapopulations. Given the fact that abalone are broadcast spawners[6], the spatial dimensions of such metapopulations depend on the scale of larval dispersal (Shepherd & Brown 1993). While fishing intensity has led to a low relative frequency of large patches in southern New Zealand, there are also other factors that are important in the spatial variation in paua abundance (McShane 1995). Growth, sizes and recruitment are found to vary over short distances and are influenced by wave exposure (Annala *et al.* 2001) and water movement (McShane 1995).

Given the biological facts of paua, it has been suggested that spatial management measures reflecting small local populations would be more appropriate than the current quota system geared towards large quota management areas (McShane, Mercer & Naylore 1994). The present management quota boundaries do not necessarily represent discrete paua stocks (Annala *et al*. 2001), yet stock assessment models assume that a paua stock in any specific QMS has the same growth and mortality characteristics in all parts of the stock. CPUE is

[5] *H. iris* can form very large aggregations of over 100 individuals in one patch (McShane 1995).
[6] Spawning is observed to occur annually during late summer and autumn.

assumed to be a constant proportion of a paua stock with homogeneous biology, habitat and fishing pressure within the modelled area, when biological facts and diver behaviour clearly outline the heterogeneous nature of paua clusters.

Large changes in the number of patches, and thus relative abundance, over large areas of coastline are only expected to occur over long periods of time, even with heavy fishing, since catch quotas restrain harvest (McShane 1995). Yet, average depletion might mask the disproportionate depletion of local stocks resulting in small-scale recruitment failure. Stock estimates by Annala *et al*. (2001) come with the caveat that serial depletion could cause model results to be overly optimistic. Given the heterogeneous nature of abalone, Shepherd & Brown (1993) suggest that no-take areas can play a vital role in addressing area-specific needs to meet the downfalls of broad-scale regulatory measures. When located in key areas of larval sources and places of high recruitment or rapid growth, a network of small no-take areas can be crucial to the maintenance of egg production, and thus recruitment. In fact, rotational closures and marine reserves are currently under consideration as a potential management tool to enhance recruitment in paua fisheries in southern New Zealand (Grindley 2003).

III. Data and preliminary analysis

To simulate the effects of no-take areas fine-scale spatial data are required. We need daily observations on diver-specific catch at each location and some measure of duration to compute a catch per unit of effort index. Ideally, we would also like information on the depth of paua harvest, boat characteristics, beach prices for a kg of live paua, diver-specific traits (such as experience, skill, income, age, education, etc.) and daily weather and ocean wave observations. This would provide a breadth of explanatory variables relevant to divers' daily participation and location decisions. Not all this data are available, but the most crucial data input, i.e. statistical area of catch, estimated weight of catch, fishing date, gathering duration and the number of divers per client permit, are recorded by the Ministry of Fisheries[7].

The Ministry of Fisheries provided two raw datasets. The first one spans the time period from 01 January 1997 to 01 October 2001, where data are restricted to paua catch within the statistical areas B1 to B16 of PAU 5B around Stewart Island[8]. The second data set covers the time period 01 October 2001 to 05 August 2003, where catch is broken down into the new, smaller statistical area units of P5BS01 to P5BS84 in PAU 5B[9], and where data is recorded per diver. Commercial catch is solely made up of the blackfooted paua *H. iris*. The latter data set was merged with the former by fitting the new 84 statistical areas to the original 16 areas[10].

The National Institute of Water and Atmospheric Research (NIWA) has recorded daily climatological observations at the Invercargill Airport from 01 January 1990 to 01 July 2003 on the total rainfall in mm (RAIN), the mean temperature in °C (MEAN), the daily mean sea level pressure in hPa (MEAN), the mean wind speed in metres per second (WIND) and the maximum gust speed in metres per second (GUST).

Distance is used as an approximation to the costs paua divers incur when choosing a statistical area. Given the Island's small size (65 km long and 40 km at its widest), it is reasonable to assume that all diving boats leave from and return to Halfmoon Bay within one day, as Oban is the only settlement providing accommodation, storage and safe anchoring[11].

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[7] The Ministry of Fisheries requires paua divers to fill in a daily Catch Effort and Landing Returns (CELR) form.

[8] In 1997, PAU 5B was divided into 16 statistical units (B1 - B16).

[9] In 2001, PAU 5B was further subdivided into 84 statistical areas (P5BS01 – P5BS84).

[10] Patches have to be large enough for travel distances between patches to be economically significant, and the biological parameters of paua have to be significantly different from each other to reflect the heterogeneous nature of diver behaviour.

[11] Personal conversation with paua divers confirmed that the island can be navigated within a few hours and that paua divers prefer to return to Halfmoon Bay by the end of the day for recreational purposes.

Similarly, divers arriving from across the Foveaux Strait usually undertake trips of several days' length and would initially anchor in Halfmoon Bay before choosing a diving patch. Figure 3 shows the geographical locations of the 16 statistical areas (Halfmoon Bay is located within the inlet by Bullers Point).

The midpoint of the mouth to the inlet by Bullers Point is selected as the starting point. Distances are measured on the map between the midpoints of the statistical areas tracing the coastal outline of the island. The 'switching point' lies between area B4 and B5, 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[12].

The two raw datasets covering the time periods 01 January 1997 to 01 October 2001 and 01 October 2001 to 01 July 2003 originally contained 1,928 and 1,356 observations, respectively. In merging the two datasets, several issues had to be addressed. Logical inconsistencies in the data revealed that many divers filled in separate forms for the same trip, and in some instances adjacent patches had been visited within one trip. Significant adjustments had to be made to reflect decreased costs by visiting adjacent patches within one trip as well as the real number of trips undertaken by the various clients, i.e. paua quota holders[13]. The final combined dataset contains 2,648 observations on client trips made to any particular patch on any day between 01 January 1997 and 01 July 2003.

[12] Surface distance comparisons (given the latitude/longitude readings) between points above and below Stewart Island, as well as to the right and to the left show that map distortions are too small to have any significant impact on the measurement of distances.

[13] Paua quota holders are small organisations, which employ a number of divers on their behalf, or individual divers who are self-employed.

A total of 48 quota holders were actively diving for paua. Fifteen out of those clients made fewer than 10 dives[14], which are eliminated from the data set for later estimation. Table I summarises statistical information regarding quota holders between 1997 and 2003. Overall, the maximum number of clients active throughout the year remains fairly stable at around 22 to 30 clients. Catch in each statistical area was summed over the time period of the dataset and plotted in Figure 4. The spatial behaviour of quota holders is clearly heterogenous with patches B12, B2 and B14 yielding the highest total paua catch of over 60,000 kg. B4, B13, B9 and B16 are among the least fished statistical areas, while the remaining patches lie somewhere in between. The spatial harvesting behaviour of permit holders supports the biological finding that paua tend to aggregate in clusters maintaining different levels of abundance. Note that the sizes of the individual statistical areas appear different in Figure 3, but since paua is harvested on rocky habitats to a depth of 10-15 meters, the actual 'harvestable' area is unlikely to be directly proportional to the size of the official catch reporting area[15].

Diving activity histograms for the total number of hours dived by statistical area, the total number of divers by statistical area and the total number of trips taken by statistical area, respectively, reveal strikingly similar patterns to the total amount of paua harvested in each patch (see Appendix, Figures 5, 6 and 7).

In later econometric analysis, the number of trips taken to any individual patch will serve 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

[14] Clients who have taken fewer than 10 trips over the given time period can be seen as outliers who have sold their quota after a few experimental dives or hold a very small quota reflecting the part-time nature of their participation.
[15] A continuous reef traces the outline of Stewart Island with clusters of paua scattered along it. It is difficult to predict how these clusters compare in size, but presumably the reporting areas B1 to B16 had been designed to split overall catch evenly.

highly significant coefficient (5.387) with a p-value of 0.000 and a R-square adjusted value of 0.9712. The constant term is not significantly different from zero with a p-value of 0.229.

IV. Modelling diving behaviour for PAU 5B

Bockstael & 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. It has set the framework for subsequent work focusing on the use of a random utility model (RUM) (Eales & Wilen 1986, Dupont 1993, Ward & Sutinen 1994, Holland & Sutinen 1999, Curtis & Hicks 2000, Holland 2000, Holland & Sutinen 2000, Mistiaen & Strand 2000, Smith 2001, Smith & Wilen 2003). A RUM postulates that the fisherman's problem is to select from a finite (and computationally manageable) number of fishing areas according to an index of 'attractiveness'. The attractiveness of each fishing area is defined by its profit, where it is a function of catch and prices, while costs depend on travel time and boat characteristics. A standard discrete choice model is applied to enable the comparison of expected utility within a probabilistic framework. Fishermen are assumed to maximise profits and will make the location choice that will yield the highest expected utility. The use of a random utility model lends itself well to the simulation of the effects of imposing no-take areas or marine reserves. Fishing location choice is treated as a behavioural decision that will change with the creation of closures, both in terms of the spatial allocation and magnitude of effort.

The nature of the paua fishery suits an approach close to the one Wilen & Smith (2001) adopt to model the spatial behaviour of red sea urchin divers in Northern California, where the participation and the choice site decision are modelled jointly. Morey, Shaw & Rowe (1991) point out that consumer surplus measures derived from discrete choice models do not usually include non-participation as one of the alternatives (and are thus defined as 'per-trip' consumer surplus measures). The total number of trips taken has to be estimated separately, outside the utility-maximizing framework, which often leads to inconsistent results.

We assume that on any given day between 01 January 1997 and 01 July 2003, paua divers face a series of discrete decisions that will influence the spatial distribution of effort as well as the overall level of participation. With no seasonal time constraints, divers choose on any given day whether to go diving or not. This choice will depend on a variety of factors, such as weather conditions, expected prices, expected abundance levels, the time elapsed since the last day fished and individual diver characteristics (such as the value of outside opportunities and leisure time, attitude towards risk, diver experience/skill and wealth). 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 are econometrically best addressed by a generalized extreme value distribution (i.e. a nested logit model rather than a logit model). Figure 8 visualizes the decision nodes.

Each diver (i) faces (M+1) choices on any day t, where M represents the number of diving patches. Divers also have the option of not diving. The utility of diver i from choice j (j = 1, …., M) can be defined by the random utility model (RUM). The subscript t is omitted in subsequent analysis to avoid cluttering.

$$U_{ij} = \beta' Z_{ij} + \varepsilon_{ij} \tag{1}$$

Z_{ij} is the nonstochastic or systematic part of the random utility model, i.e. it does not vary over the individual divers in the data set. ε_{ij}, on the other hand, represents the stochastic or random component, which reflects the idiosyncrasies of the individual divers (McFadden [1973])[16]. The systematic component Z_{ij} describes the attributes of the choice sets.

$$Z_{ij} = [X_{ij}, w_i] \qquad (2)$$

X_{ij} represents the choice-specific attributes, which vary across choices and possibly across individuals, i.e. travel costs and resource abundance. w_i captures diver-specific and time-specific characteristics that are constant across choices, i.e. weather conditions, diver individual traits, etc.

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 (extreme value) distribution[17]. The nested logit (NL) model then explicitly states the probability of choosing area j in the lower branch l (i.e. once the decision to dive has been made) as

$$P_{j|l} = \frac{e^{\beta' X_{j|l}}}{\sum_{j=1}^{J_l} e^{\beta' X_{j|l}}}$$

$$P_l = \frac{e^{\gamma' Z_l + \tau_l I_l}}{\sum_{l=1}^{L} e^{\gamma' Z_l + \tau_l I_l}} \qquad (3)$$

The inclusive value (I_l) for the lower branch is defined as the log of the denominator of the lower model. The lower level alternatives are aggregated into the branch level

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16 The random component varies across the individual divers and takes account of omitted independent variables that influence the choice of the decision maker but are unknown to the analyst.

17 $F(\varepsilon_{ij}) = \exp(e^{-\varepsilon_{ij}})$

composite alternative, so intuitively, the inclusive value describes the expected utility from choosing to dive.

The NL model provides a convenient way to relax the assumption of independence of irrelevant alternatives (IIA) of a multinomial logit (MNL) model, which is a direct result of the initial assumption that the disturbances are mutually independent and homoscedastic. This would have a significant implication for the spatial modelling of paua diver behaviour. It implies that the odds of visiting area j instead of not diving do not depend on the number of other sites available, their characteristics and their costs. The MNL model would thus predict a uniform participation response to closure. This, in essence, is what most biological models assume, i.e. effort is uniformly distributed and the imposition of a no-take area leads to a proportionate decrease in effort. The NL model avoids the IIA assumption by dividing alternatives into subgroups that allow the variance to differ across groups but not within, i.e. the IIA assumption holds within the group 'Dive' of site-choice but not across the choice of participation. It provides an answer to the question of whether individual divers will decide to go diving, and if so, how many trips they will make and which patches they will visit. This, in turn, allows us to evaluate the elimination of specific diving patches, and enables us to predict subsequent changes in participation decisions and redistribution of trips. Given the controversial nature of imposing marine reserves, this modelling approach provides a valuable tool in the assessment of potential impacts of no-take areas and adequate policy responses.

V. Estimation

Nested logit estimations and simulations are performed with NLOGIT (Version 3.0), an extension of LIMDEP[18], as it provides a range of modelling techniques tailored to the specific

[18] LIMDEP (Version 8.0) is an econometric software package developed by William H. Greene.

needs of discrete choice analysis. The utility of 'not diving' is normalized to zero, i.e. not diving yields no catch and hence no utility in this model[19].

X_{ij} is represented by the site-specific attributes of expected catch per unit of effort (CPUE) and travel distances to the various patches from the main port in Halfmoon Bay (DISTANCE). Under ideal circumstances, we would want to use patch-specific expected revenue and expected costs as explanatory variables. However, paua prices are not readily available and we concentrate on the monthly backward looking average of paua catch per unit of effort for each statistical area as an indicator of profitability[20]. The bulk of abalone destined for overseas export 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 lobster, to Asian markets. Similarly, the costs associated with commercial diving activity are rarely recorded, but distance gives some approximation of the costs divers incur when visiting a specific area.

Z_l describes the attributes of the choice sets, i.e. 'Not Dive' and 'Dive'. These include daily observations on the weather conditions described above, namely RAIN, MEAN, MSLP, WIND and GUST. In addition, a seasonal dummy is included indicating the winter season in the southern hemisphere from 01 March to 01 October (DSEASON). Ideally, we would want to include diver-specific traits, such as gender, income, education, diving experience, etc. into the analysis. However, the information provided by the Ministry of Fisheries is commercially sensitive and there is no way of linking client key numbers with specific individuals.

After eliminating all clients who had taken fewer than ten trips, the model was estimated for the time period 01 January 1998 to 01 July 2003, with 3,388 observations[21]. NLOGIT

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[19] In reality, the utility of not diving is not zero given the value of outside employment opportunities, recreation and/or not being exposed to rough weather conditions.

[20] Expected catch per unit of effort is calculated by dividing paua catch (kg) by the number of hours dived, averaged over each month for each statistical area.

[21] Catch and effort data are available from 01 January 1997 onwards, but data entries are very fragmented for the first year, most likely due to the initial familiarization period.

interprets the data as an indication of how frequently individual patches are visited, given the explanatory variables. Table II shows the estimation results of the NL model. At the first stage, NLOGIT displays the nesting structure with the proportions of choices made at each decision node. Out of all the choice occasions, 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%). These results are supportive of the results from the preliminary analysis in Figure 7 (appendix).

Most of the estimates are statistically significant at 5% significance level. Although the coefficients are not as directly interpretable as for a linear model, they provide a general indication 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 and MSLP both have a positive impact as they are indicators of warm weather. The constant term, RAIN and MSLP are insignificant at 5% significance level, but decrease the fit of the model when left out. Intuitively, we can explain these results as follows. Rain does not influence the riskiness or success of diving, and is also not necessarily an indicator of bad weather. MSLP, on the other hand, should have an impact. 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 force, which is the net force resulting from air moving from high to low pressure. Strong pressure gradients, which are identified by a tight packing of isobars[22], are associated with stronger winds, and the estimates of WIND

[22] Isobars are lines drawn on weather maps connecting points of equal pressure and are generated from mean sea level pressure reports.

and GUST show that it is the level of wind that has a significant influence on the decision to go diving. Similarly, the mean temperature and the season of the year play a significant role in the decision process.

The log-likelihood functions are evaluated at the unrestricted (L_U = -8369.574) and restricted (L_R = -9347.090) estimates. The null hypothesis is that all the slope coefficients in the nested logit model are zero, i.e. the restricted parameter space consists of the constant term. The value of the chi-squared statistic (= 1955.031) exceeds the critical value and we reject the joint hypothesis that the coefficients on the 10 explanatory variables are all zero (Pr[chi-squared > value] = .0000).

The measure of goodness-of-fit for discrete choice models is analogous to the conventional R^2 calculated as a likelihood ratio index LRI = $1-L_U/L_R$, but it is not indicative of the fit of the model and the focus should really be on the likelihood ratio test. The IV for the coefficient 'NOT DIVE' has been fixed at 1 because the nesting structure has a degenerate branch, i.e. the 'NOT DIVE' branch has only one single alternative, NOTGO. To identify the scaling of a degenerate branch, the IV of the NOT DIVE branch must be equal to 1 for identification purposes (Henscher & Greene 2000). 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 statistical significance of the IV for DIVE gives confidence in the correct specification of the model. The variances of the site participation choice are in fact different, and the estimation via maximizing random utility yields statistically significant and meaningful results.

Table III reports a set of descriptive statistics for the variables CPUE and DISTANCE, by statistical area. For each alternative the table lists the terms in the utility function for going diving as well as their means and standard deviations (S.D.). Values are provided for all observations and for the individuals that chose the particular area. The mean catch per unit of effort (over all observations) was extremely high for Areas 2, 1 and 6, in that order, implying that these areas have remained relatively unexploited, possibly due to higher recruitment rates and/or lower exploitation levels[23]. CPUE levels are used as an indicator of abundance (ceteris paribus) in stock assessment models, thus patch-specific levels of CPUE that do not coincide with the qualitative results of absolute catch levels might give some policy direction as to the prospective closure of selected areas. Similarly, mean and S.D. estimates of DISTANCE derived from the NLOGIT estimation provide an interesting insight. The measured distances should be exactly the same over any number of observations as each diver is assumed to leave from the same port. However, slight variations occur due to the calibration of data entries where patches had been visited subsequent to the visit of adjacent patches. While this does not seem to impact mean and S.D. measures calculated over all observations much, S.D. measures taken over the number of observations that chose the area in question show unusual variation for Areas 4 and 5. A likely interpretation is that these areas are frequently visited as part of a trip to other patches. These findings also have an implication for policy formulation.

NLOGIT calculates the elasticities by averaging the observation specific values, rather than by computing them at the sample means. Table IV shows 'own' and 'cross' value elasticities for each individual area[24]. The own elasticity of each area is positive and relatively inelastic for the attribute CPUE, e.g. a percentage increase in CPUE in Area 1 leads to a less than proportional percentage increase in the probability of Area 1 being visited (0.265 %). At the same time, the probability of any of the other areas (M-1) being visited decreases by 0.012 % for each area, and the probability of not going diving at all decreases by 0.011 % (i.e. the probability of participation increases). The cross elasticity is a measure of the degree to which the areas are substitutes or complements. The negative cross elasticities for all remaining

[23] For example, due to exposure to rougher currents, asymmetric information, etc.

[24] The cross elasticity is the same for all options within a branch reflecting the fact that the IIA assumption holds within the branch 'DIVE', i.e. all areas other than j denoted as (M-1) have the same cross elasticity with respect to j, but the cross elasticity for NOTGO w.r.t. j is different.

areas (M-1) with respect to the attribute CPUE indicate that diving patches are close substitutes for each other.

Own elasticity measures for the attribute DISTANCE show slightly more responsive values. The probabilities of calling on Areas 3, 4 and 5 are strongly responsive to a percentage increase in distance, which acts as an approximation to costs. For example, the probability of Area 4 being visited decreases by more then 1 % when distances increase by 1% implying elastic demand for this area. The cross elasticity for the remaining areas and NOTGO, however, are relatively small at 0.026 % and 0.022 %, respectively. Overall, elasticities provide a valuable indicator as to the responsiveness of diver behaviour in the sense that the probabilities of calling on particular patches as well as of the participation decision change. Decisions as to where to locate no-take areas should incorporate such information.

VI. Simulation of no-take areas

The closure of a patch can be simulated 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 patch to a very large negative number resulting in the prediction that no trips are taken to this area.

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, the closure of a number of patches has to be simulated simultaneously. A network of areas is more favourable to larval spill over effects and is thus preferred to a large single area. The results from trip proportions, descriptive statistics, elasticities and geographical location have to be interpreted so as to minimize the economic impact of effectively removing profitable fishing grounds.

Table V displays the simulation results of imposing a network of 4 no-take areas[25]. The baseline prediction in terms of the number of trips to each individual area (and its percentage share out of all trips) is computed by NLOGIT and compared to the scenario when certain areas are closed for fishing, e.g. the closure of Areas 4, 5, 13 and 16 result in the increase of the percentage share of the choice NOTGO by 3.24%, which translates into an overall reduction of effort by 109 trips. Some effort redistributes to the remaining areas, notably to Areas 11 and 12 by a change in the percentage share of 1.30 % and 1.34%, respectively.

Experimental simulations have been performed for closures of high CPUE areas, low CPUE areas, as well as combinations of both extremes across all statistical areas. 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 Area 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 (i.e. cross elasticity levels are negligible). However, Area 4 also is the furthest distance from the port and as such is elastic with respect to distance. Table III shows that Areas 4 and 5 seemed to be frequently targeted as part of one trip, thus it would be logical to assume that the closure of Area 4 would lead to a diminished demand for Area 5, mainly because of its geographical location (see Figure 3).

With Areas 4 and 5 as a starting base, 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 needs to be taken into account. The selected no-take areas should be spaced out in order to achieve the desired ‘network’ effect.

[25] 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 would become less discretionary.

The best results with respect to participation and distribution impacts are achieved for closures of Areas 4, 5, 13, 16, (Area 4, 5, 13, 1) and Areas 4, 5, 9, 16 (Area 4, 5, 9, 1)[26]. Table V in the appendix shows that a network of such no-take areas around Stewart Island would lead to a reduction in participation by only 109 to 114 trips between 1998 and 2003. Effort tends to redistribute, mainly to the areas 9, 10, 11 and 12. Undoubtedly, displaced effort might outweigh any stock enhancement benefits gained by this regulatory measure, 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 need not lead to substantial economic losses. Larval spill over effects over the longer time horizon might potentially even outweigh such economic losses, as hypothesized by Polacheck (1990). The results here clearly support Sanchirico & Wilen's (1999) claim that economic analysis is an indispensable component of simulating the effects of no-take areas, but biological information about water currents, transfer rates, recruitment rates, etc. has to be incorporated in order to make more accurate predictions.

VII. Conclusion

The Stewart Island paua industry has undergone marked declines in commercial catches recently notwithstanding extensive regulation by New Zealand's quota system. This situation could be addressed by the imposition of no-take areas as an additional management tool. Such a proposal would, no doubt, meet with opposition from the fishing industry as fishers feel that valuable diving grounds are taken away from them. Detailed data collected by the Ministry of Fisheries between 1997 and 2003 allow the estimation of a nested logit model of diver behaviour. On any given day, divers make the decision whether to go diving and if yes, which statistical area to choose. These two decision nodes are captured by weather specific variables in the first instance and by location-specific variables, namely catch per unit of effort and

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

distance, in the second. Despite the parsimonious fit, the model turns out to be a good fit with variables exhibiting sensible signs and being highly significant.

The model can be used to simulate the closure of a number of no-take areas according to the criterion of minimum economic impact. However, without biological data we cannot be sure of the benefits of such reserves and there is room for further research taking an interdisciplinary approach to arrive at more accurate policy predictions. It has been the aim of this paper to highlight the importance of economic input into the marine reserve debate when addressing the pressing problem of transcendent unsustainable exploitation.

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