

Harvesting in a pelagic fishery: The case of Northern Chile^{*}

Julio Peña-Torres^{a,**} and Michael Basch^b

^a *Centre for Environmental Technology, Imperial College of Science, Technology & Medicine, University of London, 48 Prince's Gardens, London SW7 2PE, UK*

E-mail: j.pena@ic.ac.uk

^b *Department of Economics, Universidad de Chile, Diagonal Paraguay 257, Santiago, Chile*

E-mail: mbasch@decon.facea.uchile.cl

This paper analyses the pelagic fishery of Northern Chile, estimating harvesting functions that contribute to understand why rather poor incentives to exit may predominate in pelagic fisheries, despite scarcer fish stocks. Our results show that per-vessel catch's stock sensitivity (the catch-to-biomass elasticity value) varies negatively with stock levels. Stock levels preceding a marked fall into biological overfishing would have been associated to biomass elasticities lower than the unitary value. This suggests that during catch bonanza periods, catch-per-unit-of-effort would fail to detect a rapidly declining stock trend, increasing the risk of fishing collapse. Moreover, external economies in search efforts would have reduced the incentives to exit, particularly for the smaller vessels in our sample. Finally, we find evidence of either constant or increasing marginal returns in the use of per-vessel fishing effort, which suggests that inefficiency in production has resulted from direct restrictions upon fishing effort. Overall, our findings provide consistent evidence that enhances the necessity of more efficient regulations upon harvesting in pelagic fisheries.

Keywords: Chilean pelagic fisheries, harvesting functions, panel estimation, Cobb–Douglas and Translog production functions, schooling fish

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1. Introduction

Marine industrial fisheries (MIFs) fit closely into the standard paradigm of the tragedy of the commons, where “too numerous and too small” participants tend to ex-

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^{**} Corresponding author. Currently at: School of Development Studies, University of East Anglia, U.K. E-mail: j.pena-torres@uea.ac.uk.

exploit “too much” of a common-pool resource. For pelagic fish¹ the analogy becomes even “more tragic”, as shown by the history of pelagic collapses [16,22]. A related issue deserving further analysis is why declining pelagic stocks have not led to corresponding reductions in aggregate fishing effort. If average harvesting costs are stock-dependent, one would expect that type of self-correcting adjustment in the fishery.

This paper explores empirically two features that are related to the concern above: catch’s stock sensitivity and the marginal productivity of fishing effort. We perform econometric estimations of harvesting functions that help understand why rather poor incentives to exit may sometimes prevail in marine fisheries, particularly in pelagic ones. We study the case of the pelagic fishery of Northern Chile between the mid-1980s and early 1990s; though this period featured declining catches triggering some degree of exit among smaller firms in this fishery, total fishing capacity of the operating fleet remained without great variations.

In addition to conditions favoring easy entry (e.g., low human and financial capital requirements; free access), overfishing has usually been facilitated by the permanence over time of numerous small vessels, owned in most cases by small family firms, which become a long-standing outcome of massive entry in the early stages of fisheries development. Unsurprisingly, the permanence over time of numerous small firms tends to be positively correlated with the persistence of common property. As an example, we can think of traditional small-scale agriculture and communal grazing activities, frequently facing overexploitation inefficiencies while operating under ill-defined private property rights, usually involving low-quality land plots [44]. Numerous participants, particularly when involving heterogeneous technologies, increase the costs of organizing collective action aimed at rationalizing common-pool uses and therefore reduce the chances for the emergence of private property.

There are numerous factors that could be relevant when attempting to explain the continuation over time of small firms at MIFs,² possibly other aspects as well may add understanding about causal links between scales of operation and the evolution of common property rights. However, none of these two concerns are primary targets in this paper. Instead, this paper aims at improving our understanding on how fishermen’s choice of vessel size (taken as a first-order approximation for choices regarding firm’s scale of operation) are conditioned by technological issues related to the nature of the fishing grounds under exploitation. We analyze input elasticities for fishing effort and fish biomass.

¹ In general, pelagic fisheries are more abundant but more variable than other fish stocks, e.g., demersal species. Pelagic fish tend to be short-lived and faster growing versus demersal ones, hence more vulnerable to recruitment fluctuations. They also tend to have significant migratory patterns. They are fish with darker and more oily flesh (vs. demersal ones), hence they tend to be less attractive for direct human consumption.

² To mention only a few examples: (i) government subsidies based on political considerations; (ii) efficiency and risk-sharing advantages of personalized, long-lasting and interlinked trading between fish-processors – traders and small-scale harvesters; (iii) financial penalties related to uncertain catches and sunk capital’s risk exposure; (iv) capital-rationing on small-scale operators, due to moral hazard and adverse selection problems (e.g., [20,39,40]).

We explore these issues by estimating Translog harvesting functions, at the vessel level, resorting to panel data on the industrial purse seiners fleet operating at the pelagic fishery of Northern Chile. There are no previous studies on these lines for Chilean MIFs. The estimation sample consists of annual information (1985–93) on per-vessel catch and fishing effort. Our estimations also resort to official annual stock assessments for the three most important pelagic stocks under exploitation. This is a multi-species fishery, with pilchard, anchovy and jack mackerel catches accounting for nearly 90% of total catches. Currently, it is the second most important MIF in Chile, with an average annual total catch (1985–93) of nearly 2.5 million tons.

We estimate harvesting functions that do not differentiate between species caught. This has three underlying reasons: (i) the *generalist* (multi-species) industrial fleet under study, (ii) the predominance of a reduction industry (fish meal and oil production) as a key demand sector, implying no significant price differentials between different species catches, and (iii) our intention to estimate catch's stock sensitivity to changes in the *aggregate* availability of the main fish stocks, as biological interdependencies between the main species imply non-trivial risks of misspecification in species-specific estimation exercises. Our modeling of harvesting functions considers proxies for aggregate availability of main fish stocks, and for aggregate and per-vessel fishing effort. Aggregate fishing effort aims to account for the possibility of contemporaneous technological externality effects (congestion or search externalities).

The paper proceeds as follows. Section 2 describes key features of the fishery under study. Section 3 discusses the harvesting model analyzed. Section 4 describes the estimation data. Sections 5 and 6 discuss the estimation methodology and the results obtained. Section 7 offers final remarks.

2. The pelagic fishery of Northern Chile

Peruvian waters are the northern boundary of these fishing grounds, implying some degree of transboundary migration.³ The area under study covers nearly 750 km of coastline, with most harvesting taking place (by a Chilean-owned fleet) within Chilean waters.⁴

As in the case of other pelagic fisheries [14], this one has shown historically a high degree of catchability, with the three main pelagic species sharing the characteristic of

³ According to IFOP's calculations, around 60% of the anchovy stock found in Chilean waters shows migratory patterns to Peruvian waters. The transboundary proportion of the pilchard stock seems to be much smaller (less than 10%). There is no clear evidence of transboundary migration for the jack mackerel stock.

⁴ The jack mackerel stock has migratory patterns towards open sea. However, a significant part of these migrations to international waters occur in areas farther South than the fishing grounds considered in this study. Since the mid-1970s and until the late 1980s, the ocean-going Soviet fleet had a significant share of jack mackerel catches in international waters of the Southeast Pacific. In the late 1980s, the Soviets annually caught around 1 million tons of jack mackerel in international waters close to the Chilean EEZ [13].

Table 1
Industrial fleet (estimation sample; balanced panel).

Years	Annual harvest (all species) per vessel (tons, 10 ³)	Annual fishing trips per vessel (number of trips)	Annual harvest/SC per vessel (times*)
average 1985–86	21.3	159	82
average 1992–93	10.6	138	41

Source: Own calculations based on IFOP's statistics. It considers 99 industrial purse seiners with $130 \leq SC \leq 380 \text{ m}^3$ (this accounts for 62% of total annual catches in 1985–93); *: here *SC* (storage capacity) has been transformed from m^3 to tons.

Table 2
Aggregate data.

Years	Aggregate effort (index)	Industrial fleet		Harvest (tons; 10 ⁶)		Biomass	Harvest/Biomass
		Number of vessels	SC (m ³ ; 10 ³)	Total	3 main species	(tons; 10 ⁶) 3 main species	(%) 3 main species
1985	100.0	177	43.4	3.155	3.057	17.085	17.9
1986	119.1	186	46.6	3.604	3.354	10.601	31.6
1987	121.9	193	49.5	2.345	2.212	14.280	15.5
1988	109.1	187	47.3	2.490	2.347	11.918	19.7
1989	121.0	193	51.2	3.039	2.875	10.115	28.4
1990	102.5	185	50.8	1.772	1.490	9.542	15.6
1991	99.6	182	52.2	1.733	1.377	11.624	11.8
1992	108.7	159	47.9	2.066	1.811	7.781	23.3
1993	110.8	157	46.6	1.915	1.672	7.229	23.0

Source: Own calculations based on IFOP's data.

Total Harvest: all fish species caught by industrial fleet's catches

Biomass: Sum of *economically exploitable* biomasses of jack mackerel, pilchard and anchovy (IFOP's annual stock assessments). Aggregate fishing effort = $\sum_i (t_i)(SC_i)$, where t_i are the annual fishing trips of vessel i and SC_i is i 's storage capacity.

moving together in densely populated groups, at relatively low depths.⁵ Since the start of its industrial development in the mid-1950s, this fishery's abundant catches have been mainly used for reduction. Total catches showed a steady increase throughout the 1970s and mid-1980s, with a maximum reached in 1986. Since then, persistent fishing pressure has led to a declining trend⁶ in the aggregate availability of the three main fish stocks (proxied by the sum of their total biomasses). In turn, this has led to declining catches (tables 1 and 2). Annual industrial catches in 1995–96 were 45% of the peak catch in 1986.

Pilchards have been the worst affected stock. Official (IFOP's) stock assessments indicate that the exploitable biomass of pilchards in 1992–93 was only 10%

⁵ The anchovy, for instance, has a vertical distribution that, in general, does not exceed 50 meters depth [47].

⁶ The strongest "El Niño" this century (until 1997–98), which peaked in the winter of 1982–83, is estimated to have contributed to this result.

of the maximum level that this stock achieved in the early 1980s. Pilchard catches have declined from representing 83% of regional industrial catches in 1985, to 18% in 1993. The pilchard's decline has occurred in parallel to a recovery, though with significant annual fluctuations, in the anchovy stock.⁷ In 1992–93, anchovies represented slightly more than 50% of regional industrial catches, while in the early 1980s anchovy catches were negligible. Jack mackerel catches in this region showed an increasing trend since the mid-1970s until the mid-1980s, when a decline – though less pronounced than in pilchard catches – started. The jack mackerel stock in the early 1990s has been estimated to be at nearly a third of the maximum regional level reached in 1985.

As a response to signs of declining stocks, the regulatory agency imposed a freezing policy on the fleet's storage capacity starting in 1986. Ever since, entry regulations have been combined with seasonal closures. Partly as a result of enforcing these regulations (though imperfectly⁸ [35]), and partly as an endogenous response to declining catches, aggregate fishing effort in the early 1990s has decreased from the peak levels achieved in the late 1980s, though this level of adjustment is far from the reduction experienced by fish stocks' aggregate availability in the decade under study (table 2). Since 1991 to date, the three main pelagic fish species have been declared to be in a state of *full exploitation*, which empowers fishery authorities to establish, in addition to entry restrictions, global and individual catch quotas. However, attempts to use catch quotas in Northern Chile have failed so far [36].

The number of purse seiners operating in the area reached a maximum in 1987. Since then the number of such vessels has gradually declined. However, the fleet's storage capacity has shown no great variations since the mid-1980s, reflecting an increasing substitution in favor of larger vessels. Though the most frequent storage capacity (*SC*) range corresponds to 230–380 m³ per vessel, the participation of larger vessels – some with a *SC* of up to 1,000 m³ – increased from 4.5% of the total number of vessels in 1985 to 20% in 1993. By contrast, vessels in the smaller size categories (e.g., with *SC* ≤ 180 m³) have declined from 60 in 1985 to 37 in 1993.⁹

The combined effect of multi-species interdependencies, the highly variable nature of the fish stocks, and the lack of long time-series statistics for these fishing grounds has led to a lack of robust scientific knowledge on the behavior of these stocks.¹⁰ This constrains the quality of any bioeconomic analysis that is currently feasible for these fisheries. Though we acknowledge the importance of improving

⁷ Csinke and Gumy [15] offer similar evidence for the Peruvian post-1973 anchovy/pilchard fishery; McEvoy [30] describes a similar phenomenon for the drop of the Californian pilchard stock through the 1950s; Cushing [16] and Sahrhage and Lundberk [41] offer additional evidence for pilchard/anchovy relationships off the South African coast in the early 1960s and off the Namibian coast in the late 60s.

⁸ For example, the freezing on the fleet's total tonnage has not been fully enforced (see table 2).

⁹ Nearly 2/3 of these smaller boats are owned by firms classifiable as "small" – according to fleet tonnage. (IFOP unpublished statistics.)

¹⁰ This tends to be a shared weakness with the scientific knowledge available for other pelagic fishing grounds around the world.

biological knowledge on these issues, this paper does not address the subject. Our focus is on studying the fishing incentives that are present in the actual harvesting technology.

An important feature of the fleet operating in Northern Chile is the absence of species-specialized vessels. One reason for this is the uncertain and highly variable nature of the main stocks under exploitation. In a highly fluctuating environment, a multi-species fleet is a form of insurance against uncertain cycles in individual species' availability [28,31]. A second reason is that pelagic fish in this region are mainly used for reduction, implying no significant price differentials between pelagic species.

Given the "economically similar" character of the main pelagic stocks under exploitation, the generalist character of the fleet under analysis, and the current lack of adequate scientific knowledge (growth patterns and species interdependencies) of the fishery under exploitation, we treat different fish species "as equal ones" in our estimations. Indeed, we use catch data that aggregate all species harvested, while we proxy fish stocks' aggregate availability by simply adding up total tonnage resulting from official (IFOP's) total biomass assessments for individual species.

The purpose of this simplified approach is, while reducing risks of misspecification problems in the estimations, to take into account the varying availability of the main fish stocks under exploitation. Although this approach precludes the testing of hypotheses concerning multi-species interdependencies (e.g., anchovy-pilchard relationship), it allows us to proxy and test in a simple and parsimonious way a possible stock-dependence of vessels' catch (across species) per unit of effort. A lower (higher) stock-dependence of catch per unit of effort tends to increase (reduce) the risk of fishing collapse. A weak "stock dependence" is another way of referring to a weak "marginal stock effect" [10]. A weaker (stronger) "marginal stock effect" tends to imply, *ceteris paribus*, a stronger (weaker) positive correlation between discount rates and stock depletion levels.

Testing the feature of catch's stock-dependence, even as a first-order approximation, is particularly relevant for the case of pelagic fisheries. It is frequently assumed that the schooling behavior of pelagic fish implies unit harvesting costs tending to be stock-independent (except for "very low" stock levels [11]), which increases the vulnerability of the stock to fishing effort. This is an important argument when explaining fishing collapses that have occurred elsewhere [16,22]. In the extreme case of no stock-dependence, the literature speaks of "pure" schooling behavior [4,5]. In a more general case, pelagic fisheries have often been described as implying catches with "weak" stock-dependence [11,14]. The latter has been interpreted as implying a catch-to-biomass elasticity that is positive but lower than one [23].¹¹ In the case of Chilean pelagic stocks, the authors are not aware of explicit testing on this issue.

¹¹ This interpretation is based on the assumption that the catchability coefficient is inversely related to the stock level (e.g., $q = dX^{-a}$, with X denoting stock level and d a constant parameter, see [5,14]).

3. Harvesting model

Our estimations consider a per-vessel harvesting function of the following general type:

$$H_{it} = f(E_{it}, B_t, A_t, R_t, \theta_{it}), \quad (1)$$

where H_{it} denotes total tonnage harvested (considering all fish species) by vessel i in year t , E_{it} is a proxy variable for vessel i 's use of variable inputs ("fishing effort"), B_t is a proxy variable for fish stocks' aggregate availability, A_t is a proxy for the aggregate fishing effort of the entire fleet operating during year t , R_t is a proxy for measuring regulatory shocks, and θ_{it} summarizes random (natural and man-originated) events affecting the harvesting success of vessel i in year t .

The strategy of collapsing variable input choices into a single variable has well-established roots in fishery economics, resting on the plausible assumption that input ratios tend to be relatively fixed in fishing operations (for short- or medium-term decisions). E_{it} is expected to be positively associated with H_{it} . However, per vessel harvesting is also conditioned by fixed investment in vessel's fishing capacity. This is a multi-attribute variable. Searching technology (sonar, airplane's support), engine power, fishing gears, storage capacity, and captain's idiosyncratic knowledge are some of the fixed factors contributing to explain differences in vessels' catch success. However, it is beyond our sample's information to attempt to cover these attributes.

In our estimations we proxy vessels' fixed fishing capacity by grouping individual vessels into representative size categories, according to their storage capacity. As a general norm, one would expect a significant positive correlation between the fishing potential related to the different fixed attributes in each vessel. Hence, differences in vessel size should behave as relatively good predictors of differences in other fixed fishing-power attributes. Our estimations will provide information for representative categories of vessels, each one characterizing an average range of fishing power attributes.

The expected sign for the effect of A_t on harvesting is unknown a priori. A_t could be related to two different effects: on the one hand, the increasing scarcity of the main stocks could have brought about *congestion* externalities (i.e., contemporaneous rival consumption) between competing vessels; on the other hand, the *schooling* behavior of pelagic stocks is expected to generate some degree of external economies when searching for migratory, high density, fish patches. Indeed, there is some evidence that searching effort from the more sophisticated vessels in this fishery tends to generate positive effects on smaller vessels' harvesting [34].

Based on a priori principles, one would expect H_{it} to show some degree of positive correlation with B_t . However, the significance of this correlation, its evolution with changing harvesting scales, and possible variations due to different vessel sizes, are all unknown. Concerning this correlation, it would be interesting to test marine biologists' frequent assertion that, for these species, stock density tends to decrease less than proportionally to reductions in its biomass [14]. This is usually interpreted as

a harbinger of collapse risks as biomass falls. The empirical validity of this assertion can be tested on estimations of the “catchability coefficient q ”.¹² If mean harvests (per unit of fishing effort) are positively associated to stock density, then the proposed relationship between stock biomass and stock density would imply an inverse relationship between the catchability coefficient and biomass levels: if biomass were to drop, we should expect q to increase. In a Cobb–Douglas harvesting function, for instance, this is equivalent to obtaining the estimated biomass-coefficient (i.e., the catch-biomass elasticity, in a log linear regression) positive but lower than one [23].

In terms of R_t , we want to account for the possible significance of a regulatory shock related to the 1988–89 controversies about the possibility of introducing historical-rights-allocated ITQs in these fishing grounds. One of the industry’s reactions was the attempt to build up higher historical catches. This strategy achieved momentum in 1989. We will test the significance of this effect by using a time-series dummy variable with unitary value for 1989, and zero otherwise. Other regulations did not experienced significant changes, in terms of changing binding regulations for the industry, along our sample period.

We estimate a Translog functional form for equation (1), whilst testing for the relevance of a Cobb–Douglas (CD) form. Both functions have been previously used for estimating harvesting functions in other fisheries¹³ (e.g., [4,6,23,26]).¹⁴ The CD form might be a useful first-order approximation for testing differences in the value of input elasticities across different vessel categories. However, CD functions impose well-known restrictions on the feasible technology [9,24]. The scale insensitivity imposed by a CD function on the values of input elasticities, for a given vessel category, is of special relevance for this study. The more general Translog model overcomes this restriction.

As shown by equation (1), we consider one-equation models, with H_{it} as the single endogenous variable. This approach is discussed after describing our data.

4. Data

All data was obtained from the Chilean Fisheries Development Institute (IFOP). It includes annual information (1985–93) on harvesting and fishing effort at vessel level, for the Chilean industrial fleet operating off the coasts of Northern Chile (between Arica

¹² In the notation of equation (1), this corresponds to $(H_{it}/E_{it})/B_t$.

¹³ A series of other recent empirical studies on fisheries have focused their estimation exercises on Translog cost or revenue functions (e.g., [8,42]). None of these studies refer to pelagic stocks (Salvanes and Steen aim at testing to what degree the stochastic environment affects the relative catch performance of vessels between fishing seasons in the Norwegian sealing fishery of Newfoundland; Campbell and Nicholl aim at testing to what degree purse seiners can target specific species in the Western Pacific tuna fishery).

¹⁴ Both Hannesson and Kirkley et al. consider the estimation of stochastic production frontiers. This method has been normally used to analyze technical (in)efficiency (inputs’ optimal mix) issues. Neither of these studies refer to pelagic species.

and Antofagasta), and time series consisting of IFOP's estimated annual biomasses for the three main pelagic species in this fishery. IFOP's stock assessments are based on Virtual Population Analysis (see [22]).

Catch data considers total tonnage caught (all fish species) by each individual vessel, for each year within the sample. Given our focus on catch's stock sensitivity to changes in the *aggregate* availability of the main fish stocks, we proxy fish stocks' changing abundance by an annual variable which sums up IFOP's estimated total annual biomasses (in tons) for the three main pelagic species. The aggregated biomass levels consider each species' economically exploitable stock.¹⁵ This includes recruitment and older age cohorts. Recruitment occurs at two years of age for the jack mackerel, at three years for pilchards, and at six months for anchovies. B_t denotes the resulting aggregated biomass variable for year t .

The fleet's aggregate fishing effort in year t is proxied by $A_t = \sum_i (Z_{it})(SC_i)$, where Z_{it} denotes the number of fishing trips¹⁶ of vessel i in year t , and SC_i is i 's storage capacity, measured in m^3 (constant across the years studied). The sum covers the entire fleet for each year. Ideally, one would also like to consider differences in trip duration across vessels of different sizes. Unfortunately, information on trip duration was not available to the authors. Hence, individual fishing effort is proxied by the annual number of fishing trips made by vessel i during year t (Z_{it}). Possible resulting estimation biases are discussed later.

All variables were transformed to natural logarithms. In terms of mathematical notation, from now on we shall use low case letters to denote the natural log of the corresponding variable (e.g., $x = \ln X$).

For estimation purposes we initially selected, from the total sample of industrial purse seiners, only those vessels active *throughout* the 1985–93 period (99 vessels in total). This group corresponds, on annual averages, to 62% of the fleet's total harvest and 55% of its total tonnage capacity for the sample period. We call this sample the “balanced” panel. As this estimation option has interest in itself, given its frequent use in the economic literature, we report the estimation results for this case. However, we later concentrate our analysis upon the estimation results for a corresponding “unbalanced” panel, in order to avoid possible risks of sample-selection bias.

In the balanced and unbalanced panels, the estimation sample was divided as follows. The vessels selected were divided into three size categories, depending on their storage capacity: group 1 (130–179 m^3), including 22 vessels in the balanced panel (and 54 in the unbalanced case); group 2 (180–229 m^3), including 8 vessels in the balanced panel; and group 3 (230–380 m^3), including 69 vessels in the balanced panel

¹⁵ IFOP's biomass estimations for jack mackerel and anchovy stocks considered an area larger than the fishing grounds studied in this paper. Hence, when constructing our proxy variable for the aggregate availability of fish stocks, the jack mackerel's and anchovy's biomass estimations were weighted by the ratio between catches in the Arica-Antofagasta area and catches in the entire sea area considered in IFOP's estimations.

¹⁶ This considers i 's fishing trips with and without success in obtaining catches.

(96 vessels in the unbalanced panel). The latter size category represents the fleet's *modal* vessel size. This way of grouping individual vessels is based on IFOP's [25] official classification regarding vessels' technological homogeneity.

For estimation purposes, we also experimented with alternative classifications. For example, we grouped the first two size categories. No significant changes were obtained versus the initial classification with three subpanels. Hence, to keep consistency with IFOP's classification, the results that we discuss are based on the latter case. For the sake of brevity and robustness in the exposition, the coefficients estimated for the intermediate subpanel (8 vessels in the balanced case) are not discussed.¹⁷ The estimations obtained for this group imply no qualitative change versus the relative patterns of the estimations for the other two more representative subpanels. Hence, in what follows we discuss the estimations for subpanels including vessels with storage capacities between (a) 130–179 m³, denoted as S1, and (b) between 230–380 m³, denoted as S2. The unbalanced panels for groups S1 and S2 include the full set of observations in these size categories.

We did not use unbalanced panel information that was available for vessels with $SC_i > 380$ m³. These vessels started to enter this fishery only on a gradual basis since 1985. In general, information about these vessels consisted, up to 1993, of very short time-series. Hence, the robustness of estimations for catch stock elasticities could become jeopardized. A similar risk was also valid for estimations concerning fishing effort elasticities: technological differences among vessels would have increased in a significant manner, had the estimation sample included vessels greater than the modal size S2 (some of them with storage capacity up to 1000 m³). In this case, robust estimation of fishing effort elasticities would probably call for information about other fixed-investment attributes, rather than just vessel size, conditioning vessels' fishing potential. The incorporation of bigger vessels in the estimation of harvesting functions in Northern Chile is left as a pending task, conditioned by the possibility of increasing the time-series length of the sample data and/or improving the information quality about cross-sectional differences between vessels' fishing capacity.

5. Econometric estimation: Methodology

Endogeneity. Strictly speaking, all three explanatory variables in equation (1) could eventually be modeled as endogenous variables, requiring the estimation of a simultaneous system of equations. However, given the *per-vessel* nature of our harvesting model, an assumed exogeneity for the *aggregate* variables B_t and A_t is probably less contentious than a similar assumption for per-vessel fishing effort.

In order to ascertain whether per-vessel and aggregate fishing effort can be considered “as if” they were exogenous variables, given the format of our estimation

¹⁷ Information on these results can be obtained on request from the authors. The small number of vessels in the intermediate subpanel weakens the robustness of the estimated coefficients for this group.

sample, Hausman exogeneity tests were carried out with the instrumental variable z_{it} for per vessel effort and the corresponding variable for aggregate effort a_t . To implement these tests, the variables $z_{i,t-1}$, a_{t-1} and $h_{i,t-1}$ were used as instrumental variables when testing for the exogeneity of z_{it} and a_t ; b_{t-1} and $h_{i,t-1}$ were used when testing exogeneity of b_t . The null hypothesis that per-vessel trip and aggregate effort are exogenous could not be rejected (using the unbalanced parsimonious Translog specification), either for small or large vessels in the sample ($F = 1.86$, p value = 0.16 and $F = 2.65$, p value = 0.07, respectively). The null hypothesis that the biomass proxy is exogenous was rejected with an F test equal to 208.6 and p value = 0.00 (again using unbalanced parsimonious Translog model).

These results have crucial consequences when estimating models such as (1). Although fishing effort could be an endogenous variable, depending on the time unit and aggregation scales considered in the estimation, it has proven to be independent from the error term in our sample. Hence, for our purposes per-vessel and aggregate fishing efforts can be considered as if they were both exogenous. This leaves variable B_t as the only possible endogenous variable that could be correlated to the error term. To overcome this problem, we use biomass lagged one period as an instrumental variable for B_t .

Given the caveats above, there is no need to treat our estimation problem as if it were a simultaneous equations system. This result can be further strengthened if we resort to the classical argument first put forth by Zellner et al. [48].¹⁸ Assuming that input decisions are based on profit-maximization behavior, Zellner et al. argue that if output is stochastic due to uncontrollable shocks such as weather, firms will select inputs to maximize *expected* profits (assuming risk-neutrality). However, in stochastic environments, the argument runs, entrepreneurs will most probably make nonsystematic errors (the authors speak of “managerial inertia” or random “human errors”) in their attempt to adjust inputs to satisfy the necessary conditions for profit maximization. When these random “human errors” are not correlated with the stochastic shocks from Nature in the production function, Zellner et al. prove that the estimation of a production function by classical least squares yields *consistent* estimators.

Stationarity. It is advisable to verify whether the variables in our estimation are stationary in time in order to avoid spurious regression problems. Should the variables be non-stationary, the usual t and F statistics do not follow the traditional Student and Fisher distributions. In this case it would be virtually impossible to make any sense with respect to the statistical significance of the estimated parameters. To check for this issue, we performed unit-root tests similar to those of Dickey and Fuller [18,19] but now correcting for the presence of cross-sections, in line with the results of Pesaran and Smith [38] and Abuaf and Jorion [1]. The appendix shows the results. The four

¹⁸ For example, [26] resort to this argument in order to justify treating per-vessel fishing effort as an exogenous variable in their estimation of a per-vessel catch (per trip) function.

variables in our estimation sample are stochastically stationary with high significance values; however, all of them show a marked deterministic trend.

As regards the latter, the trend present in each variable should be eliminated by either regressing the variable against time and rescuing the residuals or by explicitly including the time variable in the regressions to be considered. In this paper we follow the former option.¹⁹ Therefore, all following estimation results report correlations between variables which are measured as deviations versus their corresponding time trends.

Harvesting functional form. We start by estimating the following Translog model:

$$h_{it} = \alpha_{0i} + \alpha_1 z_{it} + \alpha_2 b_{t-1} + \alpha_3 a_t + \alpha_4 z_{it}^2 + \alpha_5 b_{t-1}^2 + \alpha_6 a_t^2 + \alpha_7 (z_{it})(b_{t-1}) + \alpha_8 (z_{it})(a_t) + \alpha_9 (b_{t-1})(a_t) + \alpha_{10} D89 + \alpha_{11} (D89)(z_{it}) + \mu_{it}, \quad (2)$$

where $D89$ is a dummy variable measuring regulatory shock (equal to 1 for 1989, and 0 otherwise), with all remaining variables being expressed as natural logs: z_{it} is the proxy variable for measuring vessel i 's fishing effort at time t , b_{t-1} is the instrumental variable for contemporaneous aggregated biomass, and a_t is the proxy variable for the possibility of a contemporaneous technological externality; with μ_{it} denoting the stochastic error term associated with per-vessel harvesting, and α_{0i} (a *fixed-effects* parameter) and α_j ($j = 1, \dots, 11$) the parameters subject to estimation.

Model (2) is defined in a panel format. The latter implies some convenient features. First, by using only time-series, multicollinearity problems can appear throughout, making the interpretation and measurement of individual parameters cumbersome. This pitfall is diminished by using panel samples. Second, given that we will not study expression (2) as part of a simultaneous equations system, possible identification problems and resulting simultaneity biases are avoided to a large extent. By adding cross-sectional data, these problems can be considerably lessened [27,21,2]. In our case, this seems to be confirmed by the results from the Hausman exogeneity tests. We additionally reduce the risk of simultaneity biases by introducing an adequate instrumental variable for the biomass variable.

We consider three subpanels (see section 4), all of which are estimated simultaneously using three extra dummy variables,²⁰ in order to improve on the efficiency of the estimated parameters.

Notwithstanding the merits of a panel data estimation, precision problems in the estimated parameters can arise. This can be due to idiosyncratic differences among the vessels in our sample. In addition to individual trips, it would be desirable to consider other differentiating variables (e.g., engine power, searching technology).

¹⁹ The results obtained when estimating the harvesting functions using both alternatives do not differ qualitatively.

²⁰ Three dummies were defined, one for each subpanel. $D(i) = 1$ for all data belonging to subpanel i ($i = 1, 2, 3$) and 0 otherwise. Basically all three dummies are added in our Translog model allowing us to estimate it for the entire data set at the same time in a similar fashion as in Zellner's SUR methodology.

Table 3
Translog harvesting function (unbalanced panels).

Explanatory variable	General model		Parsimonious model	
	S1	S2	S1	S2
<i>z</i>	-22.54 (-1.19)	-0.139 (0.009)	-	-
<i>b</i> (-1)	711.39 (7.91)	545.53 (12.88)	727.23 (8.03)	544.82 (13.68)
<i>a</i>	-958.65 (-6.85)	-1,358.95 (-18.03)	-983.55 (-6.76)	-1,334.34 (-18.33)
<i>z</i> ²	-0.05 (-1.27)	0.03 (0.60)	-	0.12 (23.53)
<i>b</i> (-1) ²	-11.07 (-7.28)	-7.59 (-11.72)	-12.04 (-7.71)	-7.52 (-12.28)
<i>a</i> ²	42.16 (10.12)	53.11 (25.83)	42.41 (9.63)	52.43 (26.58)
<i>zb</i> (-1)	0.77 (1.24)	-0.09 (-0.19)	0.07 (15.25)	-
<i>za</i>	0.71 (0.94)	-0.16 (0.19)	-	-
<i>ab</i> (-1)	-21.87 (-7.68)	-18.44 (-11.67)	-20.63 (-6.85)	-18.58 (-13.14)
<i>zD</i> 89	-0.13 (-4.07)	-0.02 (-0.69)	-0.16 (-11.32)	-
<i>D</i> 89	-0.07 (-0.44)	-0.44 (-2.11)	-	-0.57 (-16.77)
<i>F</i>	175.0	282.0	166.9	379.48
<i>R</i> ²	0.92	0.87	0.92	0.85
<i>n</i>	374	780	374	780
Root <i>MSE</i>	0.235	0.173	0.246	0.182

Note: *t*-statistics in parentheses, *n* is the number of observations, *F* is Fisher’s statistic, Root *MSE* is the square root of the mean square error, S1: vessels (*n* = 54) with a storage capacity (*SC*) between 130 and 179 m³; S2: vessels (*n* = 96) with *SC* between 230 and 380 m³.

Unfortunately, this information was not available to the authors. Nonetheless, the methodology used lessens this problem by resorting to a “fixed effects” model. This consists in making explicit the dependence of coefficient α_{0i} on vessel *i*. The other coefficients (α_j , $j = 1, \dots, 13$) are assumed to be constant for all vessels, within a given subpanel, throughout the period under study.

Estimating (2) by means of a “fixed effects” panel model we obtain consistent estimators for all parameters (if μ_{it} is normal, then the estimators are also asymptotically efficient). Furthermore, equation (2) was estimated using generalized least squares, by resorting to a White’s [46] heteroscedastic consistent covariance matrix. There was no need to use the stricter Newey–West procedure [33], since no autocorrelation was

Table 4
Translog harvesting function (balanced panels).

Explanatory variable	General model		Parsimonious model	
	S1	S2	S1	S2
z	21.38 (1.27)	8.67 (0.48)	–	–
$b(-1)$	163.84 (1.67)	204.44 (5.25)	142.05 (6.21)	279.68 (12.14)
a	-311.57 (-1.74)	-72.79 (-1.29)	–	–
z^2	0.004 (0.059)	0.09 (2.46)	–	0.079 (3.42)
$b(-1)^2$	-4.73 (-3.20)	-3.10 (-4.82)	-4.23 (-6.17)	-4.23 (-12.66)
a^2	10.38 (1.82)	5.46 (3.10)	0.04 (5.49)	4.67 (9.19)
$zb(-1)$	0.054 (0.15)	-0.93 (-2.01)	–	0.028 (2.32)
za	-1.37 (-1.64)	0.46 (0.52)	0.05 (11.79)	–
$ab(-1)$	-0.32 (-0.095)	-6.05 (-4.50)	–	-8.77 (-9.17)
$zD89$	0.37 (2.58)	-0.07 (-0.62)	–	–
$D89$	-2.19 (-2.67)	0.46 (0.70)	-0.13 (-2.53)	–
F	318.8	352.74	147.4	147.4
R^2	0.78	0.85	0.77	0.77
n	198	621	198	621
Root MSE	0.191	0.136	0.193	0.193

Note: t -statistics in parentheses, n is the number of observations, F is Fisher's statistic, Root MSE is the square root of the mean square error, S1: vessels ($n = 22$) with storage capacity (SC) between 130 and 179 m^3 ; S2: vessels ($n = 69$) with SC between 230 and 380 m^3 .

found.²¹ Tables 3 and 4 report the results for subpanels S1 and S2, for our balanced as well as unbalanced panel samples.

6. Estimation results and discussion

The parsimonious model results from eliminating all non-significant coefficients, testing this with a succession of Wald tests. To obtain the parsimonious model, we used

²¹ This result applies to all the estimated harvesting functions. The rejection of the null hypothesis corresponding to the presence of autocorrelation is based on results similar to the traditional DW tests but now considering the presence of cross-sections using the tables provided by Bhargava et al. [3].

Hendry's general-to-specific principle (as in [17]). Wald tests were unable to reject the null hypotheses that $\alpha_1 = \alpha_4 = \alpha_8 = \alpha_{10} = 0$ for unbalanced subpanel S1 (p value = 0.41 and F test of 1.01) and $\alpha_1 = \alpha_7 = \alpha_8 = \alpha_{11} = 0$ for unbalanced subpanel S2 (p value = 0.09 and F test of 2.02).²² Likewise, in a similar fashion, Wald tests significantly rejected the traditional Cobb–Douglas model ($\alpha_4 = \alpha_5 = \alpha_6 = \alpha_7 = \alpha_8 = \alpha_9 = \alpha_{11} = 0$) for unbalanced subpanel S1 (p value = 0.000 and $F(11, 53) = 24.81$) and for unbalanced subpanel S2 (p value = 0.000 and $F(11, 95) = 199.05$).

Comparing subpanels S1 and S2, within the general Translog model context as in (2) and taking into account the unbalanced subpanels, we confirm our sample selection criterion, in the sense that the estimated coefficients differ from each other. This tends to confirm our prior assumption of systematic differences between representative categories of vessels size.²³ This result captures differences arising from different levels of investment in fixed factors affecting fishing capacity and the resulting scale of use of variable inputs. Our approach of differentiating between representative categories of vessel size can be thought of as an alternative to the explicit measurement of the n -dimensional vector of fixed factors conditioning vessel's fishing capacity. In some contexts, our approach can provide important informational advantages.²⁴ Systematic differentiating characteristics among vessels (apart from our proxy for fishing effort), within a given size category, are captured in the estimations obtained for the vessel-specific constants α_{0i} . However, the null hypotheses that these idiosyncratic differences across individual vessels, within each subpanel S1 and S2, are equal to each other, cannot be rejected in either case.²⁵

All the estimated coefficients in the Parsimonious Translog model are strongly significant. Let us consider the resulting catch elasticities for per-vessel fishing effort, aggregate fishing effort, and aggregated biomass (unbalanced panels). The Parsimonious Translog model implies that point estimates of these elasticities are sensitive to prevailing scales of fishing effort and biomass availability. Table 5 shows point elasticity values calculated on the basis of annual averages for biomass and per-vessel and aggregate fishing effort.

The implied point elasticity values illustrate a couple of interesting patterns that seem to be robust to different ways of calibrating average values for the variables involved in the calculation of point elasticities.²⁶ Consider first the catch elasticity of

²² Similar results are available from the authors upon request for the balanced subpanels S1 and S2.

²³ For this confirmation a Chow type test was used for subpanels S1 and S2 in the unbalanced case (similar results are obtained for balanced panels). The F test (equal to 9.2) clearly rejects the hypothesis that the coefficients are equal. For the parsimonious translog models this results is, naturally, greatly enhanced.

²⁴ Access to detailed data on fleet's operations is not easily available. These problems are all the more serious, the less developed a country is.

²⁵ For the parsimonious translog model, in the unbalanced case, the F tests were 0.43 and 0.79 for subpanels S1 and S2, respectively.

²⁶ Nonetheless, our use of de-trended variables, when performing the estimation exercises, would suggest that annual averages of de-trended variables would offer a better first-round approximation when illustrating point elasticity values.

Table 5
Input elasticities (Parsimonious Translog model; unbalanced panel samples).

Years	Per-vessel fishing effort		Biomass		Aggregate fishing effort	
	S1	S2	S1	S2	S1	S2
1985	1.11	1.19	3.1	3.4	-6.5	-10.5
1986	1.12	1.22	-2.6	-1.2	7.2	7.0
1987	1.09	1.21	6.3	4.3	16.9	16.3
1988	1.11	1.19	-0.5	0.6	0.8	-1.0
1989	0.94	1.21	-0.3	0.05	12.2	12.4
1990	1.1	1.18	4.9	4.2	0.3	-2.9
1991	1.1	1.17	4.8	4.3	-1.8	-5.4
1992	1.12	1.21	-3.5	-1.4	0.27	-0.9
1993	1.1	1.22	3.7	3.0	8.5	7.1
Average 1985-93	1.09	1.22	1.6	1.8	4.6	2.9

The values for each elasticity and subpanel consider the annual averages of the variables measured in a de-trended way (and expressed as natural logs).

per-vessel fishing effort. Yearly point estimates tend to suggest in a fairly consistent way, both for S1 and for S2, that harvesting in these size categories was not subject, within the sample period, to declining marginal returns in the use of variable inputs (i.e., fishing effort); thereby suggesting the predominance of either constant (group S1) or increasing (group S2) marginal productivity for per-vessel fishing effort. The hypothesis of increasing marginal returns within the modal group S2 cannot be rejected (using the 1985-93 average point estimate, the alternative hypothesis is rejected with p value of 0.00).

If properly measured, increasing marginal returns from fishing effort would imply strong incentives for using the maximum attainable fishing effort²⁷ (given regulatory constraints) with vessels in group S2, insofar as profits could be obtained from the fishery. If true, this would also imply incentives to invest in larger vessels. This would be consistent with the type of capital replacement that has been occurring in Northern Chile over the last decade (smaller vessels being replaced by increasingly larger ones). However, we must avoid jumping too quickly to conclusions. Our measurement of fishing effort is partial. This can produce a measurement bias in our estimations of fishing effort elasticities.

Fishing trips and possible measurement bias. What type of measurement bias may possibly arise from using the number of fishing trips as a proxy for the use of variable inputs? Suppose the use of variable inputs can be summarized in total fuel use and total man-hours. The scale of use of these inputs in a given vessel depends on the level of fixed factors associated to the vessel's fishing capacity. For the sake of simplicity,

²⁷ Price-taking behavior in input and output markets is the expected behavior in this fishery. Also, the existence of positive switching costs (i.e., the cost of turning heating furnaces on and off in fish meal processing) seems to preclude the private optimality of pulse fishing (e.g., [10]).

assume that differences in fixed fishing capacity among vessels are properly captured by our representative categories of vessel size. Vessel i 's total use of fuel in year t (F_{it}) and of total man-hours (L_{it}) can be written as

$$F_{it} = T_{it} \cdot d_{it} \cdot f_{it}, \quad L_{it} = T_{it} \cdot d_{it} \cdot l_{it}, \quad (3)$$

where T denotes number of fishing trips, d the average trip duration (hours per trip), f the fuel used per hour, and l the crew's labor effort per hour. Assume that F and L denote the total level of harvesting services rendered by the use of fuel and labor. Both T and d are scaling factors of the input services provided per unit of time. Separate input elasticities could be defined for fuel use and labor effort. In either case, the input elasticity would represent the ratio between percentage changes in catch and the sum of percentage changes in each of the three components of F and L . When the use of variable inputs is proxied by the scaling factors T or Td , the measurement of a fishing effort elasticity represents the sum of the underlying elasticities for fuel use and labor effort.

To characterize the possible measurement bias in our estimations of fishing effort elasticities, suppose that we leave aside the possibility of technological change in fuel use and labor (i.e., assume that both f and l are constant across the years studied, as well as across different vessels within a given size category). In this case, the measurement bias in fishing effort elasticities estimated by using T (instead of Td), within a given category of vessel size, depends on the ratio $(\Delta d/\Delta T)$, where Δ denotes percentage changes across years and vessels, within a given category of vessel size. If percentage changes in the omitted variable d_{it} were "small" relative to the percentage variation in T_{it} , our estimations could be considered as an adequate measurement of the underlying (added) input elasticities for fuel and labor. A similar rationale applies when considering the possibility of technological change, i.e., the ratios $(\Delta f/\Delta T)$ and $(\Delta l/\Delta T)$ would add to a possible measurement bias.

Currently we have no empirical information to assess the possibility of technological change in f or l . This is an interesting challenge left open for future research. Hence, our estimations of fishing effort elasticities must be interpreted as based on the premise that no significant technological improvement occurred along the vessels considered in the estimation sample.

What about the possible measurement bias resulting from the lack of information about d_{it} ? Although information on trips' duration, at the vessel level, was not available to us, we did obtain information on trips' annual average duration, for each category of vessel size, during 1990–94 (IFOP). In this period, fishing trips of vessels in S1 had an average duration of 20 hours, whereas trips of vessels in S2 lasted on average 25 hours. In both groups, annual deviations from these averages were quite small (the standard deviation is 0.9 in S1 and 2.6 in S2). This may be an indication that average trip duration has not changed too much in this fishery along the sample period. It could well be that this is related to the fact that we are studying a fleet specialized in coastline harvesting.

The argument above is only conjectural, as we have no information on the levels of variation in d across individual vessels, for each of the size categories considered in our estimations. Nonetheless, our estimations of individual effort elasticities provide useful information as long as they are interpreted as assuming that (i) trip duration did not change significantly (relative to ΔT) along time and across vessels, within a given size category, and (ii) the possibility of technological change in the amount of services rendered by fuel use and labor effort, per unit of time, is not considered.

The bottom-line of the proposal above is that the surprising result²⁸ of increasing returns in fishing effort for vessels in the modal group S2 must be considered as a provisional finding. The important implications from possible increasing returns from marginal fishing effort in the modal vessel size (i.e., greater fishing pressure on fish stocks; also probable productive inefficiencies as a result of regulations on fishing effort and, hence, policy trade-offs between productive efficiency and conservation targets when choosing regulatory instruments) imply that further research on this feature is needed.²⁹ A greater emphasis on the use of cross-sectional data would probably help to reduce problems when recruiting the required information. The task of measuring individual fishing effort in a more robust way is left for future research.

Per-vessel catch's stock dependence. Yearly point estimates of catch-to-biomass elasticities show important variations, both in terms of the involved sign as with respect to the resulting absolute values. The latter are significantly affected by the way we proxy average values for the variables affecting the calculation of the point elasticities. These point estimations must be interpreted with caution: firstly, because point elasticities represent – by definition – local concepts, whereas our estimation sample includes a range of biomass variations which are far from involving “marginal” changes; secondly, because our estimation methodology (i.e., use of de-trended variables) reduces the role played by time-series effects in the estimation procedures. However, despite the implied uncertainties upon the specific point elasticity values involved, a couple of results should be interpreted as robust readings.

Firstly, our estimation results show that variations in aggregate biomass availability have had a statistically significant effect upon per-vessel catch along the period under analysis. In the case of pelagic fish stocks, this is not an obvious result, as previously discussed. This result is conditioned by the fact that, along the period studied,

²⁸ Surprising in the sense that, in principle, it goes against standard economic theory. However, different arguments could be made to explain why the modal vessel size may show increasing returns in marginal fishing effort. For example, fishers might not achieve optimal scales of operation due to regulations impeding optimal levels, or composition, of capital replacement or fishing effort. This is the argument that Bjørndal [4] uses to justify his finding of increasing returns in per-boat marginal fishing effort in the North Sea herring fishery. Bjørndal's estimation results do not differ qualitatively when proxying per-boat fishing effort by number of boat-days at sea, versus proxying it by number of fishing trips per boat.

²⁹ Other empirical studies on *pelagic* fisheries have also found evidence in this direction [6,4]; though one must not rush to conclusions as these studies consider different data aggregation and different weights between time-series and cross-sectional data.

the aggregate biomass availability fell significantly (more than a half in less than a decade).

Secondly, yearly point elasticity estimates suggest that smaller vessels' (group S1) catch may be more sensitive to variations in biomass availability, versus the case of the modal group S2. One may conjecture that this result reflects different capacities to adjust fishing effort to changes in the spatial distribution of the more productive fish patches. Larger vessels might be less vulnerable to fish stock variations, due to their greater maneuvering capacity and search capabilities, whereas smaller vessels' catch would be more quickly affected as fish stocks become scarcer. This would be consistent with the observed gradual exit of smaller vessels in Northern Chile since the mid 1980s, when this fishery started its declining trend towards biological overexploitation.

Thirdly, it can be argued that yearly variations in the point estimates of catch-to-biomass elasticities do not provide a fully consistent picture in terms of the involved sign in the correlation between these two variables (particularly when calculating point estimates on the basis of averages of de-trended variables). However, one should realize that yearly variations in the estimated point biomass elasticities are affected by simultaneous changes in the variables that condition the calculation of these point estimates. One way to isolate the "pure" biomass effect, i.e., keeping constant all other relevant effects, is to calculate point estimates of biomass elasticities which are function only of biomass levels, for example by considering 1985–93 averages for the value of per-vessel and aggregate fishing effort. Using averages of de-trended variables, figure 1 plots the resulting point biomass elasticities, as a function only of aggregate biomass levels.

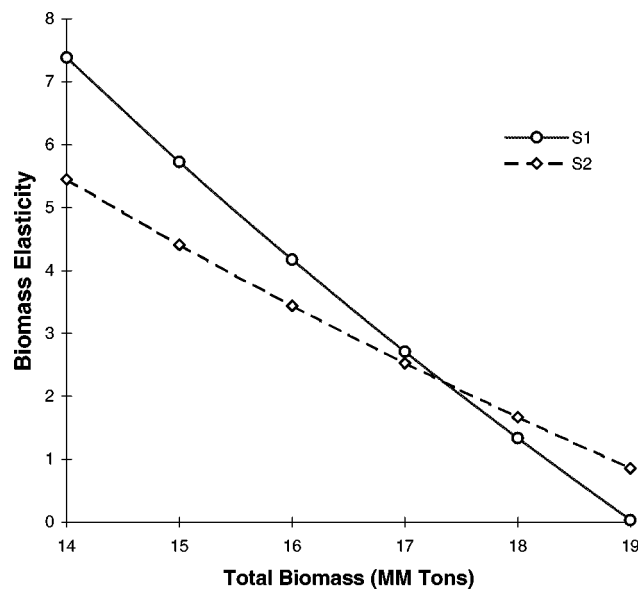


Figure 1. Biomass elasticity (Translog model unbalanced panels).

Following this approach, the positive value of the “pure biomass effect” catch-to-stock elasticity becomes now clear, at least for the range of aggregate biomass levels observed along the period under study. Another interesting and robust reading, which appear to be fully in line with marine biologists’ theories [14], is the negative sign obtained for the relationship between the value of the catch-to-stock elasticity and prevailing stock levels: greater abundance of these pelagic stocks seems to be associated with lower levels of per-vessel catch’s stock-sensitivity.

The divergence (from above) of the estimated point biomass elasticities, versus a unitary-value elasticity, increases as biomass levels fall further behind those estimated as prevalent during the first half of the 1980s (see table 2 and [35]). It was during this latter period that a sustained trend of declining aggregate biomass started at this fishery. Hence, our estimation results are consistent with the conjectural proposition that a “relatively weak” stock sensitivity of per-vessel catch (i.e., biomass elasticities lower than one³⁰) could have prevailed throughout the period of booming catches in the pelagic fishery of Northern Chile, thereby reducing the incentives to slow down fishing effort. Once the fishery entered a stage of biological overexploitation (this probably started along 1988–89), the stock sensitivity of catch per unit of effort would have started to increase. However, we cannot be certain about the magnitudes involved in the latter effect.

In order to provide for more robust readings about stock-specific values for per-vessel catch’s stock sensitivity, one would need to introduce biologically-related structure into the estimation model. This is not a trivial task. It is well known that environmental shocks have significant, if not deciding, effects upon the relationship that may exist between recruitment and parental stock in this fishery. This discussion, however, escapes the scope of this paper.

Contemporaneous externalities from aggregate fishing effort. Yearly variations in the sign of the point estimates of the aggregate-effort elasticity reflect the fact that, in the Parsimonious Translog model, the value of this elasticity is scale-dependent with respect to biomass availability and the level of aggregate fishing effort. According to our estimations, a positive sign for this elasticity is more likely the scarcer fish stocks are, and the higher aggregate fishing effort become. This could be interpreted as evidence supporting the prevalence of positive search externalities under this type of fishery context. If true, this could be indicating external economies in the search for scarce and highly mobile fish patches.

The yearly point estimates of the aggregate-effort elasticity are positive more frequently for group S1 than for group S2. This may suggest that vessels in the smaller size category would benefit from external search economies to a greater extent than larger vessels, as they may enjoy a stronger degree of free-riding from aggregate

³⁰ Recall that this feature would support the frequent presumption that, for pelagic stocks, fish stock density tends to decrease less than proportionally vis a vis reductions in its biomass. Under this feature, catch per unit of effort would fail to detect a rapidly declining stock, increasing the risk of stock collapse.

search effort. This seems to be consistent with evidence of a quick propagation of private (i.e., individual vessel's) information regarding the location of productive fish patches in this fishery [34, chapter 3].

A predominance of positive search externalities, for the more frequently observed levels of aggregate biomass and aggregate fishing effort along our sample period, would help understand why smaller vessels in Northern Chile have not been substituted at a faster pace by vessels with greater fishing capacity³¹ (given our preliminary finding of increasing returns in fishing effort from vessels in group S2). A better understanding of this issue is important, as it is related to the permanence of "too numerous" insiders exploiting the commons. Smaller boats tend to be predominant among small firms. However, larger firms have also maintained a proportion of relatively small boats in their fleets. It could well be that search externalities are enhanced by a larger number of vessels in operation, for a given level of total fishing capacity. This conjecture is probably more valid, the more randomly distributed the fish patches are. Another reason might be that the chances of undetected non-compliance with existing regulations may increase by operating with a larger fleet, as one would expect enforcement costs to be positively correlated with the number of vessels operating.

A more efficient way to internalize search externalities could be one of the reasons underlying the not so unusual feature of industrial concentration in pelagic fisheries, including concentration in harvesting sectors.³² So far, not much attention has been devoted to issues of industry structure in fisheries, with the exception of discussions regarding concentration in processing sectors and the effect of monopsonistic power upon harvesters [12,32,43,45]. Further research on these issues could offer useful insights to the management of industrial fisheries.

7. Final remarks and policy implications

This paper uses panel data to estimate Translog harvesting functions for the pelagic fishery of Northern Chile. The results are differentiated according to representative cohorts of vessel size. The estimated values of catch-to-input elasticities result being scale-dependent. We obtain robust evidence that a greater abundance of pelagic stocks tends to be associated to lower levels of per-vessel catch's stock sensitivity. Consistent with this, stock levels preceding a marked fall into biological overfishing, which probably started taking real shape around the mid-1980s, would be associated to biomass elasticities lower than the unitary value. If valid, this conjecture would suggest that during bonanza periods, catch per unit of effort is not a good predictor of

³¹ This would add to arguments already mentioned in footnote 2. Regulatory restrictions upon capital replacement could be another important reason.

³² Peña [34] describes cases of pelagic fisheries subject to industrial concentration, both in processing and harvesting sectors: the Peruvian anchovy fishery during the 1960s and early 1970s, and the Chilean (Northern and Southern) pelagic fisheries, since the mid 1970s to current times. Manning [29] and Peña [37] offer insights on the observed industrial concentration in the case of the Namibian pilchard fishery.

the start of a declining stock trend. The estimated negative relationship between the biomass elasticity value and prevailing stock levels would imply that, once the fishery has entered a stage of biological overfishing, less productive vessels will have some incentive to exit the fishery. Nonetheless, positive search externalities, along several of the years under study, would have reduced the incentives to exit, particularly for the smaller vessels in our sample. Finally, smaller vessels' fishing effort seems to be governed by *constant* marginal harvesting returns, while larger (modal-size) vessels display *increasing* returns in marginal fishing effort. The latter is a surprising result that may involve a measurement bias. Future research is expected to focus on this issue.

On the whole, the results above provide consistent evidence that helps understand why aggregate fishing effort in this fishery failed to adjust rapidly enough to avoid a significant fall in aggregate biomass (more than halving it in less than a decade). Similar stories have previously occurred in other pelagic fisheries around the world. The initial abundance of these stocks has been source for substantial profits in the past, with several fishing collapses resulting from it. Overall, this provides indisputable evidence about the vulnerability of pelagic stocks to unrestrained fishing.

This evidence has important policy implications. In Northern Chile, the fishery's authority has been unable to overcome industry's opposition to catch quota proposals. Substitute regulation has usually been only imperfectly enforced. Similar controversies currently surround regulatory decisions in the pelagic fishery of Southern Chile (with jack mackerel catches of 3.5 mill. tons in 1996, and a recently proposed quota of 1.8 million tons for 1999, after the 1997/98 El-Niño years). In the North, fishing entrepreneurs have made successful use of two instrumental arguments (to stop introduction of more exacting regulations). They have argued that (i) cyclical biological substitution among the main caught species implies that reductions in a given single stock will be offset by increases in the stocks of competing species, allowing for continuity in fishing activities; and that (ii) the significant role played by environmental shocks, particularly upon the relationship between recruitment and parental stock, makes the use of catch quotas economically inappropriate [34].

In terms of the entrepreneurs' first argument, the evidence accumulated since the mid-1980s and the results from our estimations imply that the proposed offsetting effects on fishing activities have flaws. Aggregate biomass availability has been significantly affected by initial reductions in the pilchard stock, not for only two or three years, but for more than a decade. This has had significant effects upon harvesting. Our estimations also suggest that the effects upon catch per unit of effort, in a period of booming catches but rapidly falling stocks, do not provide enough "brake incentives", upon fishing effort, to self-stabilize the fishery. This means that *binding* regulations upon fishing effort do have an important role to play. Our preliminary finding of increasing marginal harvesting returns for the modal vessel size, if it overcame a more robust measurement, would imply that direct restrictions upon fishing effort have resulted in tradeoffs between conservation targets and the pursuing of productive efficiency. If this were true, the promotion of a fleet composed of larger but fewer

vessels would probably facilitate policy tradeoffs. However, political reality may lead to the contrary. In this case, policy choices would be tougher.

This paper provides no new evidence about the entrepreneurs' second proposition. However, the general principle involved in this argument is disputable. This proposition is related to well-known controversies in marine biology: "For stocks that are highly sensitive to environmental changes... it is difficult to establish that (fishing mortality induced) reductions in the adult stock will reduce recruitment... until the process has gone so far that recruitment is seriously affected" [14, p. 297]. However, the uncertainties involved, when assessing the relationship between recruitment and parental stock, are not an indisputable barrier for implementing cost effective quota regulations. Simulation-tested management options can be a cost effective mean for assessing the involved tradeoffs between expected catch returns and risk to the fish stock. The post-1994 management of the South African anchovy/pilchard fishery is a valuable example of this approach [7].

Some extensions of the analysis in this paper would contribute to test the robustness of the policy insights suggested here. For example, a more exact measurement of fishing effort, as well as attempts to differentiate between species-specific biomass elasticities. In Northern Chile, the latter would require studying issues related to an expected (though still not well understood) correlation between anchovy and pilchard stocks. In general, a more thorough modeling of time-series effects would contribute to strengthen the robustness of the analysis. Also, accumulated years over the period studied in this paper facilitate the incorporation of bigger vessels into the estimation exercise. From a more general perspective, now also extending the argument to other industrial fisheries, the empirical study of technological features affecting incentives to adjust fishing effort, as stock levels vary, is a research area that can offer fruitful policy insights.

Appendix: Augmented Dickey–Fuller tests

Let x_{it} be any of the following variables (considering log transformation): per-vessel harvest, aggregate fish stock, per-vessel fishing effort (proxied by the number of fishing trips) and aggregate fishing effort. We estimate the following regression by least squares which includes both a constant and a temporal tendency:

$$\Delta x_{it} = \alpha + \beta t + \gamma x_{i,t-1} + \sum_{k=1}^n \delta_k \Delta x_{i,t-k} + \varepsilon_{it}. \quad (4)$$

Here, $\Delta x_{it} = (x_{it} - x_{i,t-1})$, i denoting the vessel and t the year; α , β , γ and δ_k are parameters that have to be estimated, ε_{it} are white noise errors, and n represents the minimum number of terms of the type $\Delta x_{i,t-k}$ that have to appear in (4) so that the error term is white noise.

The appropriate test is similar to the augmented Dickey–Fuller test (ADF). The usual t and F statistics are matched with critical values established numerically via

Table 6
ADF tests.

Variable	β	γ	\overline{R}^2	DW	F	n
Log Z_{it}	-4.16	-7.07 (0.00)	0.05	1.89	29.5	1,189
Log H_{it}	-2.76	-6.13 (0.00)	0.03	1.70	19.1	1,189
Log B_t	-3.41	-4.01 (0.01)	0.64	2.37	8.0	9
Log A_t	-1.71	-3.52 (0.02)	0.61	2.21	6.6	8

Notes: The null hypotheses to be verified are: (i) $\gamma \geq 0$, which means that variable x in (4) is stochastically non-stationary, and (ii) $\beta = 0$ which means that there is no tendency. The coefficients corresponding to β and γ are the normal t statistics, which are matched against the critical values of Abuaf and Jorion. The figures in parentheses correspond to the p values of these tests. \overline{R}^2 is the adjusted R^2 coefficient; DW is the Durbin–Watson statistic, F the Fisher statistic, and n the number of observations.

Monte Carlo simulations [1]. In expression (4), parameter γ measures possible stochastic non-stationarity of x_{it} , whereas coefficient β measures the deterministic non-stationarity or tendency of such a variable. Table 6 shows the results of the pseudo-ADF test for each of our estimation variables.

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