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**Supply response in
fisheries - the North Sea**

S Pascoe and S Mardle

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Centre for the Economics and Management of Aquatic Resources (CEMARE), Department of Economics, University of Portsmouth, Locksway Road, Portsmouth PO4 8JF, United Kingdom.

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Abstract

Supply response in fisheries is largely masked by environmental factors. Hence estimating elasticities of supply is difficult from official landings data. In this paper, a bioeconomic model of the North Sea fishery is used to generate estimates of international landings of several key species at different price levels. By using a model to estimate the landings, the stochastic element is removed from the data. Short and long run elasticities of supply for cod, haddock, saithe, whiting, sole, plaice and nephrops are estimated from the derived data.

Introduction

In most competitive industries, the quantities supplied would be expected to increase with an increase in price. The responsiveness of quantity supplied to changes in price can usually be estimated through econometric analysis using market data (taking account of the effects of change in quantity supplied on price itself through the demand relationship).

In fisheries, estimating the supply response is less straight forward. The level of catch is generally taken as a function of the combination of inputs employed and the level of stock (see for example, Hannesson 1983, 1993a). Stock size varies in density both spatially and temporally. As fish stocks are a mobile resource, the stock density varies from place to place and day to day. In addition, the fish stocks are often influenced by environmental conditions (e.g. rainfall, water temperature, tides and currents), exhibiting seasonal migratory patterns as well as seasonal aggregation and dispersion. As a result, the ability of fishers to increase supply in response to changes in price may be affected by the relative abundance of the species available for harvest. As a result, the supply has a large stochastic element.

While seasonal variables can be included in econometric models to allow for some of these factors, the relationship between supply of landings and price may be further masked by other factors. Many species occupy the same habitat, resulting in a mixed bag of catch. Hence, landings of a species may not decrease if its price decreases if it is caught in association with a higher valued species or the prices of the other species in the catch increases. With different fishers operating in different areas using different gear (and hence experiencing different catch combinations), any supply response is difficult to extract from the available landings data.

These factors are generally considered to dominate the supply of individual species. In most demand studies involving fish, supply has been assumed to be exogenously determined (e.g. Bird 1986; Ioannidis and Whitmarsh 1987a, 1987b, Pascoe, *et al.* 1987, Jørgensen 1988, Barten and Bettendorf 1989, Bjørndal, Salvanes and Andreassen 1992, Kirman 1992, Bjørndal, Gordon, and Singh 1993, Gordon, Salvanes and Atkins 1993, Herrmann, Mittelhammer and Lin 1993, Wessells and Wilen 1993; Cooper and Whitmarsh 1994, Asche 1996, Bose and McIlgorm 1996, Jaffry, Pascoe and Robinson 1997).

Many fisheries are also subject to quota control. This places an upper limit on the amount of landings. Even though catch of a species may still occur, this over quota catch is discarded. Hence, it would be expected that the official recorded landings would not increase even if price increased if the total allowable catch had already been achieved.

In this paper, an attempt is made to estimate elasticities of supply for the key quota species caught in the North Sea. A bioeconomic model of the fishery (Mardle *et al* 1997) is used to estimate the short and long term level of landings of the key species given different price levels. An advantage of this approach is that environmental fluctuations and other factors that affect local abundance will not affect the derived data. As a result, much of the stochastic element is removed.

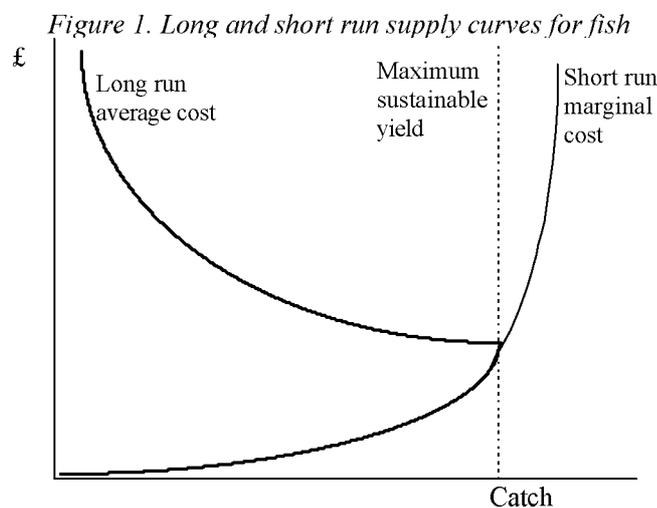
The short and long run supply of fish

As with any industry, the quantity of a species landed depends on the cost of catching it and the price received. The level of catch depends on the stock abundance and the level of fishing “effort” (a composite measure of fisheries inputs). Hence, the cost per unit of catch is a function of the cost of fishing effort and the stock abundance.

In the short run, fisheries, like most industries, are subject to the law of diminishing returns. Catch per unit of effort is often highest at low levels of effort, and diminishes with higher levels of effort. Stocks are generally not evenly distributed along the sea bed, and in many cases the areas of higher abundance are fished first. As fishing is primarily a hunting activity, fishermen need to search for the fish once the areas of known abundance have been exploited to their potential. Lack of perfect knowledge as to other’s activities results in some fishermen applying effort to areas that have already been fished out. Similarly, lack of perfect knowledge about the location of the remaining resource results in effort being applied to unproductive areas. Since the probability of finding additional fish decreases with increased catch, the catch per unit of effort must also decrease.

As a result, the marginal cost per unit of catch is likely to increase with increased landings. As there is only a finite amount of fish available, the marginal cost increases exponentially (Figure 1). As noted above, the stock size is affected by a range of factors. Many of these factors are stochastic in nature (e.g. climatic conditions). As a result, the short term supply curve shifts continuously, making the econometric estimation of supply difficult.

In the longer term, the stock size is also affected by the previous levels of harvest. If the catch exceeds the sustainable level, the stock size decreases in subsequent years. Copes (1970) suggested that the long run “supply” curve for fish would be backward bending, based on the average cost (rather than marginal cost) of fishing. At high price levels, the fishery is likely to be able to sustain high level of efforts resulting in low levels of sustainable catch. Conversely, at low prices the fishery is likely to sustain only low levels of effort, again resulting in low levels of catch. In the long run, the catch cannot exceed the maximum sustainable yield.



Empirical estimation of the short run fisheries supply curve is made difficult through the high degree of stochasticity arising from environmental factors. To account for these effects, accurate indexes of abundance are required. While these are generally estimated by fisheries biologists for managed species at an annual level, they are not generally estimated at a more disaggregated level. The estimation of long run supply curves from landings data is also difficult as the fishery is never in an equilibrium position. While dynamic models can be applied, these again need an appropriate index of stock abundance for each period.

No empirical analyses of supply response have appeared in the literature, most likely due to the problems raised above. Instead, supply response has focused on the responsiveness of effort production to changes in profitability (e.g. Bjørndal and Conrad 1987, Hannesson 1993b, Ye and Beddington 1996, Yew and Heaps 1996) and on the relationship between the level of effort and the level of catch (e.g. Agnello and Anderson 1981, Hannesson 1993a, Pascoe and Robinson 1998). Greenberg, Herrmann and McCracken (1995) examined the allocation of Alaskan Snow Crab to different markets (given a quantity supplied) to assess its responsiveness to price. However the total quantity supplied was assumed to be exogenously determined (i.e. not responsive to price).

The data problems facing the econometric analysis of supply response can be overcome by using a bioeconomic model to estimate the level of landings given different prices. Bioeconomic models provide a means of combining what is known about the biology and the fleet into a single framework for policy analysis. The models may be used to estimate how a fleet would respond to changes in price given its cost structure and a given stock abundance. Hence the stochastic stock effects are removed and short run elasticities estimated. Population dynamics can be incorporated into the model to estimate the long run levels of landings given various prices, allowing estimates of the long run elasticities of supply to be made.

Similar approaches have been used to estimate the supply response in other industries from model derived data. Binkley (1993) used a forestry bioeconomic model to estimate the long run price elasticity of supply for timber. In agriculture, Fulginiti and Perrin (1993) used a model of a tobacco farm to estimate the supply response in the absence of quotas. The derived data were used to estimate the own price elasticity of supply. Similarly, Horbalk (1993) used a partial equilibrium model to estimate the own price elasticity of supply for cattle farms while Kingwell (1994) used a programming model of a wheat farming system to estimate the supply response with different levels of risk aversion.

The North Sea Fishery

The North Sea (ICES divisions IIa and IV) is a multi-species multi-gear fishery, of great importance to many countries. The North Sea is the major fishing grounds for most species caught in European Community waters. The total value of the catch in 1994 was estimated to be about ECU 750 million. Over half of the combined total allowable catches of all species in all EU waters is from the North Sea. Commercial activity in the region is mostly undertaken by fishers from the countries bordering the North Sea: UK, Denmark, The Netherlands, France, Germany, Belgium and Norway.

The fishery is managed according to the guidelines of the CFP as each is a member of the EU, except Norway who co-operates with the defining of suitable management measures. At the time of inception, the method of quota definition amongst the member states was based on three main factors: historic catch, compensation for loss of catches in EEZs and sensitive fishing regions. Similarly today, North Sea TACs are assigned on the basis of recent historic data.

The fishing activity relevant to human consumption is concentrated on eight species; cod, haddock, whiting, saithe, plaice, sole, nephrops and herring. The first seven species are demersal species (i.e. bottom dwelling) whereas herring is predominantly a pelagic species (i.e. surface dwelling). Hence, the fishing operation for herring is different than that of the other species. All of these species have yearly TACs imposed by the EC. The roundfish stocks of cod, haddock and whiting are heavily fished with approximately 60% of their biomass removed each year, making recruitment very important. Cod, plaice and herring are currently considered to be overexploited and at risk of collapse. Stocks of the other demersal species are below the level that produces the maximum sustainable yield. The species are dependent on each other, with considerable interaction in the food chain (ICES 1996a).

The distribution of the seven main demersal species to the eight main countries is shown in Table 1. Each country's catch share is given as their assigned TAC in 1995 (ICES 1996b).

Table 1: TAC by country and species in 1995 (in tonnes).

	Cod	Haddock	Whiting	Saithe	Plaice	Sole	Nephrops
Belgium	4560	930	1707	160	6610	2459	795
Denmark	23260	6360	7030	4340	16210	1700	795
France	4780	7070	11830	25314	660	500	25
Germany	11780	4050	1960	12331	6670	1831	110
Netherlands	12750	510	4370	15	50860	22192	410
Norway ^a	8500	2500	100	50000	600	300	200
UK	50960	68030	30093	8590	25940	1318	13065

^a The 'TAC' for Norway is estimated from 1994 landings.

Attempts at bioeconomic modelling in the North Sea have been limited. Kim (1983) developed a surplus production multispecies model of the demersal fishery to estimate the potential economic rent that could be achieved. Two alternative regimes were investigated: one with an economic objective and another with an biologic objective.

Bjørndal and Conrad (1987) and Bjørndal (1988) developed a model of the North Sea herring fishery that included a fleet dynamics function where entry or exit depended on the sign of normalised profit per boat. That is, if profits were positive then boats would enter, whereas if profits were negative then boats would leave. While the model allowed for changes in the fleet size, it did not allow for changes in the fleet structure. The model examined the dynamics of the fishery as it approached the open access level of effort.

Frost *et al* (1993) developed two bioeconomic models of the North Sea fishery. A linear programming model was used to estimate the optimal allocation of effort of Danish trawlers, from two ports, between three fishing areas. A larger simulation model was used to estimate levels of effort and catches for eight countries by species and gear type. Unfortunately neither model incorporated stock nor fleet dynamics.

Dol (1996) developed a simulation model of the flatfish (sole and plaice) fishery in the North Sea. The model focused primarily on the Dutch beam trawl fleet, and was used to estimate the potential benefit of an area closure for plaice.

Mardle *et al* (1997) developed a multi-objective programming model of the north sea fisheries to estimate the long run optimal level of effort given economic, employment and biological objectives. The model included both population dynamics as well as predator prey relationships (where appropriate). This model was subsequently modified to include both long run and short run yield curves (Mardle and Pascoe 1997) to examine trade-offs between long and short run objectives. This latter model forms the basis of the model used in this analysis, details of which will be presented below.

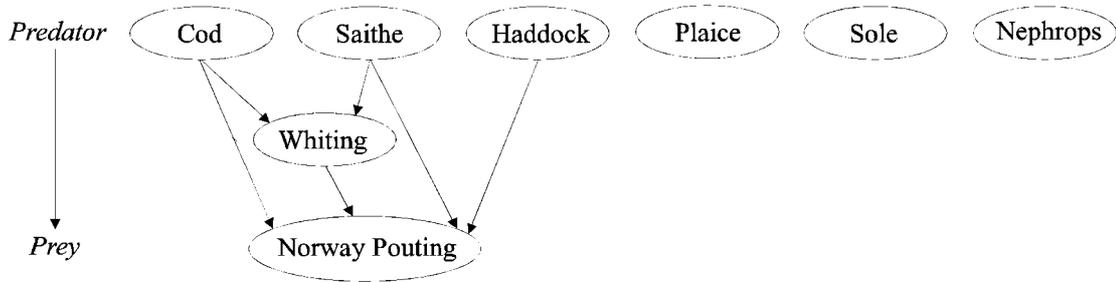
North Sea demersal fishery bioeconomic model

The North Sea demersal fishery bioeconomic model contains both a short run and long run component model. In the long run component, the estimated species' biomass and catch are in an equilibrium state. In the short term, the yield may exceed the long term sustainable level. The stock dynamics are developed using multispecies logistic growth models of the form in equation (1).

$$G_i = r_i B_i \left(1 - \frac{B_i}{\sum_{s_i \in S_i} \beta_{s_i} B_{s_i} + K_i} \right) - \sum_{s_j \in S_j} \alpha_{s_j} B_{s_j} \quad (1)$$

where G_i is the growth of species i , r_i is the growth rate, K_i is the environmental carrying capacity (excluding the effects of the modelled prey species), B_i is the biomass, $s_i \in S_i$ is the set of predator species and similarly $s_j \in S_j$ is the set of prey species. The predator-prey interactions between the species included in the model are shown in figure 1. The biomass of Norway Pouting which is a prey of four of the species is assumed constant, as this species is not explicitly incorporated into the model.

Figure 1: Species' predator/prey relationships.



Nonlinear regression analysis was used to estimate the parameters for the growth models from the annual catch, effort and biomass data given in ICES (1996b). Details on these regressions are presented in Appendix 1.

The structure of the bioeconomic model considers the seven most important demersal species i in the North Sea, includes the North Sea's seven coastal states j (eight by managing England and Scotland individually), and takes account of the four associated major fishing methods or gear types k . The key variables are estimated for each of,

- species (i): cod, haddock, whiting, saithe, plaice, sole and nephrops;
- countries (j): Belgium, Denmark, England, France, Germany, Netherlands, Norway and Scotland; and
- gear types (k): otter trawl, seine, beam trawl and nephrops trawl.

Price per tonne of fish landed is variable, using price flexibilities, based on Jaffry, Pascoe and Robinson (1997), to estimate the effect of changes in the level of landing on price. The average price of each species in each country in 1995 was estimated and used as the base. Fixed costs and running costs of vessels by country and gear type were estimated from 1995 statistics (Concerted Action on Fisheries Economics 1997). Also, crew wages were taken as a proportion of the revenue achieved (Frost et al. 1993). The number of boats present in the fishery by country and gear type, and their respective days at sea, were similarly obtained (Concerted Action on Fisheries Economics 1997). Gear selectivity by species and gear type was taken also from the North Sea simulation model of Frost et al. (1993). Differences in catch rates by boats from different countries were estimated as a scaling factor by comparing derived catch from observed catch. Here, catch was assumed to be a linear function of effort

(defined in terms of days fished standardised using the scaling factor above), gear selectivity and biomass. The equilibrium biomass was estimated as a function of fishing effort, while the short term biomass was based on information reported by ICES (1996b). New boats were acceptable for countries with existing boats containing a gear type, and a landings limit of 400 tonnes of fish per year is imposed on all boats.

All of the species currently have yearly TACs assigned with a historically proportional divisions to the relevant countries. In this model, England and Scotland are treated independently of each other, where the proportion of UK TAC assigned to each is estimated in proportion to current boat numbers.

The mathematical representation of the model with variable and parameter descriptions is given in Appendix 2. The parameters in the model come from a variety of sources, some of which may not be comparable. In a number of cases parameters were not available for some species and/or countries. In this case, estimates based on comparisons with other countries were used. Therefore, the results of the model need to be viewed as indicative rather than predictive.

Model simulations and results

The model was used to estimate the total European short run and long run catch of the key species with different price levels. The model was run 300 times with the base price for each species varied stochastically. The base price was multiplied by a factor to simulate the effects of an exogenous shift in demand. A uniform distribution ranging from 0.25 to 1.75 was assumed for the demand shift factor, resulting in an index of prices. The objective function used in the model was the maximisation of the short term gross margins, subject to the constraint that the resultant level of effort was sustainable (at least at the open access equilibrium level) in the longer term. The TAC restrictions were relaxed to allow the effects of this constraint to be removed from the model results. However, while the global TAC was relaxed, the maximum share of the catch of each country was still restricted in line with the Common Fisheries Policy Principle of Relative Stability (Holden 1994).

Log linear supply curves were estimated using OLS regression, with the log of landings the dependent variable and the log of the prices index as the independent variables. From the 300 derived observations, catch of at least one species was zero in 5 cases resulting in only 295 effective observations. As the prices were randomly generated, there were no problems of multicollinearity (as would most likely occur from market data). Similarly, as the data were not derived from a time series there were no problems of autocorrelation or non-stationarity. Other factors that may affect supply, such as fishing costs and environmental factors, were all constant in the model.

The regression results are presented in Appendix 3. The regression models for cod, saithe and haddock were generally considered to be good, with adjusted R^2 values of the short term models in excess of 0.7. For the other species, the goodness of fit measure for the short term models was considerable lower - in the order of 0.3 for whiting, plaice and sole and 0.17 for nephrops. For sole and nephrops, the adjusted R^2 was considerably greater for the longer term model than the short term model (around 0.5).

The own and cross price elasticities of supply are summarised in Tables 2 and 3. In general, supply was generally inelastic in the short term. In most cases, a one per cent change in price would result in a less than 0.1 per cent change in the quantity supplied. The exception to this was haddock and saithe, with own price elasticities of supply estimated to be around 0.9 and 0.5 respectively. The own price elasticities tended to be larger in the long term than in the short term. For cod, the short term own price elasticity was not significantly different from zero, but was estimated to be about 0.1 in the long term.

Table 2. Estimated own and cross price short run elasticities of supply

Species landed	Price						
	Cod	Haddock	Saithe	Whiting	Plaice	Sole	Nephrops
Cod			-0.04	-0.04	-0.10	-0.22	
Haddock		0.88	-0.13			-0.36	-0.09
Saithe	0.14	-0.06	0.53	0.07			0.04
Whiting	-0.10	0.09	0.05	0.06	0.04	-0.04	
Plaice	-0.03	-0.05		-0.07	-0.07	-0.09	
Sole	-0.08	-0.08	-0.08		-0.16		
Nephrops	0.11	-0.30	0.10				0.25

As the fishery is a multi-species fishery, the catch is generally comprised of a number of species. The ability of fishers to target individual species is limited, reducing the incentive to switch from species to species in response to price. However, some switching of gear types may occur as a result of price changes. In the short term, an increase in sole price is expected to result in a decrease in the landing of cod as some fishers change activities. In contrast, an increase in cod, whiting or nephrops price is estimated to result in an increase in saithe landings as saithe is generally caught as bycatch with these species. The converse relationship, was also observed for whiting and nephrops. As saithe is a relatively low valued species compared with cod, increase in its price were not sufficient to encourage increased landings of cod.

In the results obtained, not all short term own price elasticities made economic sense. The own and cross price elasticities for plaice were all negative, implying that a price increase in any species would result in a reduction in the landings of plaice. This result (combined with the low R^2 value) may suggest that the functional form of the model is not appropriate. A more flexible functional form (such as the translog) which allows variable elasticities may be more appropriate for this (and other) species.

The long run elasticities were generally larger, and in many cases the opposite sign to the short run elasticities. For example, an increase in the price of sole is likely to lead to a short term reduction in the landings of cod, but an increase in the longer term. This is because the lower landings of cod in the short term results in a higher cod biomass, resulting in a higher catch per unit of effort in the long term.

Table 3. Estimated own and cross price long run elasticities of supply

Species landed	Price						
	Cod	Haddock	Saithe	Whiting	Plaice	Sole	Nephrops
Cod	0.11	-0.15	-0.11	0.03	-0.03	0.13	
Haddock		0.86	-0.13			-0.30	-0.08
Saithe	0.02		0.50	0.08	0.02	0.03	0.04
Whiting	0.35	-0.37	-0.20	0.17		0.40	
Plaice	0.04	0.11	0.14	0.07	0.25	0.18	
Sole		0.02			0.02	0.03	
Nephrops		-0.53		-0.16	0.14		0.35

The effects of the predator prey interactions are also apparent in the long run elasticities. An increase in the price of cod results in an increase in the landings of cod in the long term. As cod is a predator of whiting, the higher landings results in a lower cod biomass, increasing the biomass of the whiting stock.

Discussion and Conclusions

The model of the North Sea demersal fishery was developed in order to assess the effects of changes in economic conditions (e.g. prices and costs) and different management policies on the long term structure and profitability of the fishery. In this paper, the model was used to estimate the long and short run supply responsiveness to price.

The model is currently based on a number of parameters from different sources, some of which may not be comparable. In a number of cases, parameters had to be estimated based on secondary information. As a consequence, the results presented in the paper are indicative rather than absolute. Nevertheless, the results of the model suggest that, in the short term, supply of the key species is generally inelastic. In the longer term, the population dynamics and predator prey relationships also affect the supply response.

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Appendix 1. Biological component of North Sea model

The biological component of the model is based on the concept of surplus production. This is the growth in the biomass that can be harvested on a sustainable basis. The growth of a species depends on the level of its own biomass. For predator species, the growth also depends on the level of biomass of the prey species while for prey species the growth depends on the level of biomass of the predator species.

The level of growth of each species was estimated as a function of its own biomass and the biomass of predator/prey species. The data used were the estimated biomass of each species over the last 20 years (ICES 1996b). The functional forms that provided the best fit are presented below. Whilst a number of parameters appear to be not significantly different from zero, excluding them significantly reduced the predictive power of the model (i.e. the R^2). Equilibrium biomasses were estimated by equating the growth to the linear catch function $C = qEB$, where q is the catchability coefficient, E is the effort and B is the biomass. For the following species i , the growth equation is denoted by G_i and similarly the equilibrium biomass equation is denoted by B_i .

Note that the biomass of Pout is denoted by B_p .

Cod (c)

$$G_c = r_c B_c \left(1 - \frac{B_c}{\alpha_1 B_w + \alpha_2 B_p + K_c} \right) \quad R^2 = 0.211$$

$$B_c = \left(\alpha_1 B_w + \alpha_2 B_p + K_c \right) \left(1 - \frac{q_c E_c}{r_c} \right)$$

Coefficient	Value	Standard error	t-statistic
r_c	0.5078	0.2669	1.903
α_1	0.6513	1.7661	0.369
α_2	0.3096	0.5983	0.517
K_c	77.30	1240.26	0.062
F = 3.49 (5 %)			

Haddock (h)

$$G_h = r_h B_h \left(1 - \frac{B_h}{\beta_1 B_p + K_h} \right) \quad R^2 = 0.334$$

$$B_h = \left(\beta_1 B_p + K_h \right) \left(1 - \frac{q_h E_h}{r_h} \right)$$

Coefficient	Value	Standard error	t-statistic
r_h	0.8188	0.2960	2.766
β_1	0.2448	0.1022	2.395
K_h	578.755	271.607	2.131
F = 8.44 (1 %)			

Whiting (w)

$$G_w = r_w B_w \left(1 - \frac{B_w - \gamma_1 B_c - \gamma_2 B_s}{\gamma_3 B_p + K_w} \right) \quad R^2 = 0.451$$

$$B_w = (\gamma_3 B_p + K_w) \left(1 - \frac{q_w E_w}{r_w} \right) - \gamma_1 B_c - \gamma_2 B_s$$

Coefficient	Value	Standard error	t-statistic
r_w	0.0791	0.2523	0.319
γ_1	0.0020	0.0140	0.143
γ_2	0.2394	0.3665	0.653
γ_3	1.3782	0.3982	3.461
K_w	88.59	300.31	0.295
F = 8.10 (1 %)			

Saithe (s)

$$G_s = r_s B_s \ln \left(\frac{B_s}{\lambda_1 B_p + K_s} \right) \quad R^2 = 0.282$$

$$B_s = (\lambda_1 B_p + K_s) \left(1 - \frac{q_s E_s}{r_s} \right)$$

Coefficient	Value	Standard error	t-statistic
r_h	0.4046	0.0987	4.099
λ_1	0.1297	0.1427	0.909
K_h	1018.84	596.67	1.708
F = 15.167 (1 %)			

Sole (so)

$$G_{so} = r_{so} B_{so} \left(1 - \frac{B_{so}}{K_{so}} \right) \quad R^2 = 0.434$$

$$B_{so} = K_{so} \left(1 - \frac{q_{so} E_{so}}{r_{so}} \right)$$

Coefficient	Value	Standard error	t-statistic
r_{so}	0.5376	0.0945	5.689
K_{so}	224.94	62.48	3.600
F = 26.55 (1 %)			

Plaice (pl)

$$G_{pl} = r_{pl} B_{pl} \left(1 - \frac{B_{pl}}{K_{pl}} \right) \quad R^2 = 0.308$$

$$B_{pl} = K_{pl} \left(1 - \frac{q_{pl} E_{pl}}{r_{pl}} \right)$$

Coefficient	Value	Standard error	t-statistic
r_{pl}	0.3748	0.1826	2.053
K_{pl}	1680.03	1706.00	0.985
F = 20.47 (1 %)			

The biomass for pout (B_p) was important for most species. As it was not incorporated directly in the model, an average biomass was assumed for the purposes of the analysis. Adequate data was not available to develop similar models on nephrops. However, estimates of carrying capacity and growth rate were made based on current levels of catch and effort.

For most of the above species, the growth models explained less than 40 per cent of the variation in growth. While such a low explanatory power would be considered inappropriate for the purposes of setting actual TACs, the models were considered adequate for the purpose of demonstrating the potential of the MOP technique.

Appendix 2. Mathematical description of the base model

$$\begin{aligned} \min z = & w_{L1} \frac{npL}{Max\ ProfL} + w_{L2} \sum_j \frac{ne_j + pe_j}{TB_j} + w_{L3} \sum_i \frac{pdL_i}{sTAC_i} + w_{L4} \sum_j \sum_i \frac{ntL_{ji} + ptL_{ji}}{TAC_{ji}} + \\ & w_{S1} \frac{npS}{Max\ ProfS} + w_{S2} \sum_j \frac{ne_j + pe_j}{TB_j} + w_{S3} \sum_i \frac{pdS_i}{sTAC_i} + w_{S4} \sum_j \sum_i \frac{ntS_{ji} + ptS_{ji}}{TAC_{ji}} \end{aligned} \quad (6)$$

subject to,

$$\sum_j \sum_k f_{profL_{jk}} + npL - ppL = Max\ ProfL \quad (7a)$$

$$\sum_j \sum_k f_{profS_{jk}} + npS - ppS = Max\ ProfS \quad (7b)$$

$$\sum_k Emp_{jk} boats_{jk} + ne_j + pe_j = \sum_k Emp_{jk} NB_{jk} \quad , \forall j \quad (8)$$

$$\sum_j \sum_k (catchL_{jki} - landL_{jki}) + ndL_i - pdL_i = 0 \quad , \forall i \quad (9a)$$

$$\sum_j \sum_k (catchS_{jki} - landS_{jki}) + ndS_i - pdS_i = 0 \quad , \forall i \quad (9b)$$

$$\sum_k landL_{jki} + ntL_{ji} - ptL_{ji} = TShare_{ji} ntac_i \quad , \forall i, j \quad (10a)$$

$$\sum_k landS_{jki} + ntS_{ji} - ptS_{ji} = TAC_{ji} \quad , \forall i, j \quad (10b)$$

$$ntac_i \geq \sum_j \sum_k landL_{jki} \quad , \forall i \quad (11a)$$

$$ntac_i \leq \sum_j \sum_k catchL_{jki} \quad , \forall i \quad (11b)$$

$$days_{jk} \leq SeaDays_{jk} boats_{jk} \quad , \forall j, k \quad (12)$$

$$catchL_{jki} = (Select_{ik} Scale_{ji}) days_{jk} bio_i \quad , \forall j, k, i \quad (13a)$$

$$catchS_{jki} = (Select_{ik} Scale_{ji}) days_{jk} Biomass_i \quad , \forall j, k, i \quad (13b)$$

$$f_i = \sum_j \sum_k (Select_{ik} Scale_{ji}) days_{jk} \quad , \forall i \quad (14)$$

$$bio_c = (0.651 bio_w + 0.310 BioPout + 77.302)(1 - f_c / 0.508) \quad (15)$$

$$bio_h = (0.245 BioPout + 578.76)(1 - f_c / 0.819) \quad (16)$$

$$bio_w = (1.378 BioPout + 88.59)(1 - f_c / 0.079) + 0.002 bio_c + 0.239 bio_s \quad (17)$$

$$bio_s = (0.130 BioPout + 1018.84)(1 - f_c / 0.405) \quad (18)$$

$$bio_{so} = 224.94(1 - f_c / 0.538) \quad (19)$$

$$bio_{pl} = 1680.03(1 - f_c / 0.375) \quad (20)$$

$$bio_{ne} = 300(1 - f_c / 0.2) \quad (21)$$

$$price^*_{ji} = AvP_{ji} \left(1 - \left(PF_{ji} \frac{\sum_k (land^*_{jki}) - TAC_{ji}}{TAC_{ji}} \right) \right) \quad , \forall j, i \quad (22a/b)$$

$$frev^*_{jk} = \sum_i price^*_{ji} land^*_{jki} \quad , \forall j, k \quad (23a/b)$$

$$f\ cost^*_{jk} = FC_{jk} boats_{jk} + VC_{jk} days_{jk} + WV_{jk} frev^*_{jk} \quad , \forall j, k \quad (24a/b)$$

$$fprof^*_{jk} = frev^*_{jk} - fcost^*_{jk}, \forall j, k \quad (25a/b)$$

$$land^*_{jki} \leq catch^*_{jki}, \forall j, k, i \quad (26a/b)$$

$$\sum_i land^*_{jki} \leq 0.4boats_{jk}, \forall j, k \quad (27a/b)$$

Note: The * in equations (22a/b)-(27a/b) denotes the fact that the equation is required for the short term variable and the long term variable.

Indices

- i* Species; cod, haddock, whiting, saithe, plaice, sole and nephrops.
j Country; Belgium, Denmark, England, France, Germany, Netherlands, Norway and Scotland.
k Gear type; otter trawl, seine, beam trawl and nephrops trawl.

Variables

<i>fprof_{jk}</i>	Profit (mECU).
<i>frev_{jk}</i>	Revenue (mECU).
<i>fcost_{jk}</i>	Costs (mECU).
<i>price_{ji}</i>	Price ('000 ECU).
<i>catch_{jki}</i>	Catch ('000 tonnes).
<i>land_{jki}</i>	Landings ('000 tonnes).
<i>days_{jk}</i>	Number of days fished ('000 days).
<i>boats_{jk}</i>	Number of boats.
<i>f_i</i>	The fishing mortality, i.e. catchability ' effort.
<i>bio_i</i>	Species biomass ('000 tonnes).
<i>ntac_i</i>	Total allowable catch ('000 tonnes).
<i>np (pp)</i>	Negative (positive) deviation from the economic rent goal.
<i>ne_j (pe_j)</i>	Negative (positive) deviation from the nbr employed goal.
<i>nd_i (pd_i)</i>	Negative (positive) deviation from the discard goal.
<i>nt_{ji} (pt_{ji})</i>	Negative (positive) deviation from the TAC goal.

Data

<i>w_l</i>	Achievement function weight for l th goal (l=1,...,5).
<i>MaxProf</i>	Maximum profit achievable in the fishery (mECU).
<i>TAC_{ji}</i>	Total allowable catch ('000 tonnes).
<i>sTAC_i</i>	Total allowable catch by species ('000 tonnes).
<i>TShare_{ji}</i>	Current TAC share per country by species.
<i>NB_{jk}</i>	Current number of boats.
<i>TB_j</i>	Total number of boats by country.
<i>Emp_{jk}</i>	Current employment.
<i>AvP_i</i>	Average UK price in 1995 ('000 ECU).
<i>PF_{ji}</i>	Price flexibility coefficient.
<i>FC_{jk}</i>	Fixed cost per boat ('000 ECU).
<i>VC_{jk}</i>	Variable cost per day ('000 ECU).
<i>WC_{jk}</i>	Wages as a percentage of revenue.
<i>Select_{ik}</i>	Gear selectivity coefficient.
<i>Scale_{ji}</i>	Relative efficiency index.
<i>SeaDays_{jk}</i>	Estimated maximum number of days fishing per vessel.
<i>BioPout</i>	Constant for the biomass of Norway pouting ('000 tonnes).
<i>Biomass_i</i>	Estimated species biomass in 1995 ('000 tonnes).

Appendix 3. Regression results

Cod

Ln(Price)	Short run			Long run		
	Coefficient	Standard Error	T-statistic	Coefficient	Standard Error	T-statistic
Constant	4.1933	.0054	771.196 **	3.9914	.0082	484.053 **
Cod	.0145	.0093	1.563	.1115	.0141	7.913 **
Haddock	-.0051	.0092	-5.59	-.1479	.0140	-10.558 **
Saithe	-.0436	.0097	-4.487 **	-.1136	.0147	-7.701 **
Whiting	-.0426	.0093	-4.546 **	.0308	.0142	2.168 *
Plaice	-.0995	.0095	-10.448 **	-.0314	.0144	-2.175 *
Sole	-.2181	.0095	-22.749 **	.1259	.0145	8.660 **
Nephrops	-.0073	.0093	-.782	-.0057	.0142	-.403
R Square		.702			.522	
Adj R Square		.695			.511	
F		96.792			44.902	

* significant at 5% level; ** significant at 1% level

Haddock

Ln(Price)	Short run			Long run		
	Coefficient	Standard Error	T-statistic	Coefficient	Standard Error	T-statistic
Constant	3.2376	.0191	169.207 **	3.5802	.0165	216.784 **
Cod	-.0426	.0327	-1.304	-.0352	.0282	-1.249
Haddock	.8834	.0325	27.172 **	.8611	.0280	30.685 **
Saithe	-.1251	.0342	-3.655 **	-.1319	.0295	-4.464 **
Whiting	-.0407	.0329	-1.236	-.0378	.0284	-1.328
Plaice	-.0251	.0335	-.751	-.0304	.0289	-1.054
Sole	-.3578	.0337	-10.605 **	-.3049	.0291	-10.468 **
Nephrops	-.0859	.0329	-2.604 **	-.0789	.0284	-2.772 **
R Square		.750			.788	
Adjusted R Square		.744			.783	
F		123.462			152.855	

* significant at 5% level; ** significant at 1% level

Saithe

Ln(Price)	Short run			Long run		
	Coefficient	Standard Error	T-statistic	Coefficient	Standard Error	T-statistic
Constant	4.0890	.0087	467.528 **	4.4537	.0057	781.174 **
Cod	.1364	.0149	9.124 **	.0243	.0097	2.499 *
Haddock	-.0570	.0148	-3.839 **	-.0153	.0096	-1.583
Saithe	.5277	.0156	33.713 **	.4950	.0102	8.011 **
Whiting	.0724	.0150	4.803 **	.0787	.0098	48.510 **
Plaice	-.0266	.0153	-1.741	.0212	.0099	2.124 *
Sole	-.0244	.0154	-1.583	.0289	.0100	2.879 **
Nephrops	.0441	.0150	2.928 **	.0350	.0098	3.568 **
R Square		.81390			.89857	
Adjusted R Square		.80936			.89610	
F		179.31264			363.22164	

* significant at 5% level; ** significant at 1% level

Whiting

Ln(Price)	Short run			Long run		
	Coefficient	Standard Error	T-statistic	Coefficient	Standard Error	T-statistic
Constant	3.3516	.0088	377.239 **	2.8395	.0285	99.295 **
Cod	-.1036	.0151	-6.822 **	.3492	.0483	7.224 **
Haddock	.0905	.0150	5.996 **	-.3674	.0487	-7.539 **
Saithe	.0463	.0159	2.914 **	-.2024	.0511	-3.958 **
Whiting	.0621	.0153	4.059 **	.1684	.0501	3.361 **
Plaice	.0375	.0155	2.416 *	.0887	.0499	1.776
Sole	-.0409	.0156	-2.612 **	.4035	.0519	7.768 **
Nephrops	.0100	.0153	.656	.0137	.0491	.280
R Square		.302			.396	
Adjusted R Square		.285			.380	
F		17.767			25.597	

* significant at 5% level; ** significant at 1% level

Plaice

Ln(Price)	Short run			Long run		
	Coefficient	Standard Error	T-statistic	Coefficient	Standard Error	T-statistic
Constant	4.5768	.0063	715.992 **	4.6935	.0106	442.098 **
Cod	-.0345	.0109	-3.163 **	.0379	.0181	2.092 *
Haddock	-.0555	.0108	-5.116 **	.1073	.0180	5.951 **
Saithe	-.0208	.0110	-1.888	.1371	.0190	7.218 **
Whiting	-.0762	.0114	-6.668 **	.0688	.0183	3.761 **
Plaice	-.0686	.0111	-6.129 **	.2511	.0185	13.503 **
Sole	-.0853	.0112	-7.571 **	.1821	.0187	9.728 **
Nephrops	-.0129	.0110	-1.177	.0110	.0183	.602
R Square		.390			.570	
Adjusted R Square		.375			.559	
F		26.241			54.373	

* significant at 5% level; ** significant at 1% level

Sole

Ln(Price)	Short run			Long run		
	Coefficient	Standard Error	T-statistic	Coefficient	Standard Error	T-statistic
Constant	3.2822	.0087	374.475 **	3.3523	.0034	966.832 **
Cod	-.0766	.0149	-5.116 **	-.0084	.0059	-1.420
Haddock	-.0765	.0148	-5.142 **	.0182	.0058	3.102 **
Saithe	-.0763	.0156	-4.869 **	.0037	.0062	.608
Whiting	-.0217	.0151	-1.439	.0016	.0059	.270
Plaice	-.1569	.0153	-10.222 **	.0187	.0060	3.086 **
Sole	-.0203	.0154	-1.314	.0290	.0061	4.748 **
Nephrops	-.0160	.0151	-1.065	-.0010	.0059	-.173
R Square		.390			.129	
Adjusted R Square		.375			.108	
F		26.265			6.087	

* significant at 5% level; ** significant at 1% level

Nephrops

Ln(Price)	Short run			Long run		
	Coefficient	Standard Error	T-statistic	Coefficient	Standard Error	T-statistic
Constant	-.7049	.0288	-24.430 **	.3169	.0225	14.055 **
Cod	.1086	.0493	2.202 *	-.0669	.0385	-1.735
Haddock	-.2978	.0490	-6.074 **	-.5288	.0383	13.801 **
Saithe	.1053	.0516	2.040 *	-.0728	.0403	-1.804
Whiting	-.0569	.0497	-1.145	-.1634	.0388	-4.204 **
Plaice	-.0178	.0505	-.354	.1406	.0395	3.560 **
Sole	.0128	.0508	.252	.0337	.0397	.850
Nephrops	.2521	.0497	5.068 **	.3506	.0388	9.021 **
R Square		.187			.515	
Adjusted R Square		.167			.504	
F		9.445			43.683	

* significant at 5% level; ** significant at 1% level