The Benefits to Fisheries of UK Intertidal Salt Marsh Areas

R&D Technical Report E2-061/TR

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Statement of Use
This document provides guidance to Environment Agency staff, research contractors and external agencies involved in the process of appraisal of projects in which intertidal salt marsh is either created or lost.

Key words
Salt marsh, Production Function Models, Managed Realignment, Environmental Economics, Fisheries, Flood Defence

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EXECUTIVE SUMMARY

Background

The policy context for the project is the need to understand the ecological benefits of managed realignment projects. When sea walls are set back to a new line of defence, an area of intertidal habitat is created. One of the major potential benefits of such schemes is the value of creating such habitat for use by commercially important fisheries species at crucial stages in their lifecycle. When tidally inundated, salt marsh habitat provides refuge and a rich feeding ground to young fish and crustacea in particular. The economic quantification of this ecological value has often proved elusive. If this indirect-use benefit is to be included in a cost-benefit analysis (CBA) revenue stream, some kind of economic model is required.

Main Objectives

The objectives of the report are threefold. Firstly, the ecological and economic literature relating to the fisheries benefits of intertidal areas is reviewed. Secondly, a simple economic model is described to formalise the contribution of salt marsh area to the production function of fisheries in the area. Thirdly, the project aims to estimate the coefficients of the explanatory variables in the model for the purposes of benefit function transfer and subsequent use by practitioners of project appraisals for flood defences.

Results

Data sets derived from secondary sources were used to estimate the elasticity of production of salt marsh and fishing effort for: a) all species and fishing gear types and b) shellfish species and gear types. The results of panel econometric estimation were not significant for the salt marsh explanatory variable in both cases. The non-significance of the regression means that subsequent welfare calculations to estimate a monetary value for an increase in salt marsh area were not possible.

Conclusions/Recommendations

Two possible conclusions can be drawn from the empirical part of the project. Either the model requires re-specification with more explanatory variables but with the same kind of functional form, or the methodology is too simplistic to be able to isolate the

contribution of the salt marsh to the production function. The latter of the two is the more likely to be correct.

A bioeconomic model of harvesting, with salt marsh as a component of the growth of the biomass stock is suggested as a possible alternative approach. However, the potential for wider application in the context of benefits function transfer may be limited by the availability of the necessary data sets. A further, more large-scale and interdisciplinary study would be required in order to address this issue.

1 INTRODUCTION

1.1 Salt marshes and Flood Defences

Land claim from the sea in the UK began in Roman times when sea walls were built to enclose salt marshes for grazing (Little, 2000). Progressive enclosure has resulted in extension of these enclosed areas for various uses, from agriculture to industry. The UK currently has over 2000km of coastal flood defences and so the economic consequences of maintaining and upgrading flood defences are significant (Crooks and Turner, 1999).

The UK Climate Impacts Programme (UKCIP), suggest a “medium-high” scenario of a 2.1^{o}C warming in global temperatures by 2050. This translates to an increase in sea level globally of 28cm, which is predicted to be partially offset or exacerbated by iso-static adjustments in the natural land movements. As a consequence, intertidal zones will increasingly be subject to a process known as “coastal squeeze” (Crooks & Turner, 1999). As sea levels rise, the total area of intertidal flats becomes increasingly confined as inland encroachment is prevented by the presence of “hard defence” structures. Therefore, the desire to protect and defend valuable developments and important inland habitats from inundation by the sea is in conflict with the desire to prevent diminution of the total area of intertidal habitat.

Traditionally, government policy in the UK has favoured hard defence structures. Flood defence projects on vulnerable stretches of coastline have been funded by the Ministry of Agriculture, Fisheries and Food (MAFF, now DEFRA) on the basis of favourable project appraisal documents, showing a positive net present value (NPV). However, the cost-benefit analysis (CBA) is often incomplete with some important components of Total Economic Value (TEV) being excluded from the analysis.

Given the low productivity of agricultural land in the UK, where agricultural support inflates the value of the land, managed realignment has been considered by many to be an economically efficient and environmentally sustainable means of flood defence (Bowers, 1999; Packham & Willis, 1997).

Burd (1995) defines managed realignment as:

> *"setting back the line of actively-maintained defence to a new line inland of the original, or preferably to rising ground. In most situations, active management is carried out, aimed at establishing a viable and sustainable salt marsh, although mudflats may be desirable."*

Managed realignment may be carried out to provide a low-maintenance flood defence or solely for habitat creation. Clearly, the potential for achieving both at the same time is considerable in certain sites. The major focus of managed realignment project appraisals has been the flood attenuation benefits resulting from a reduced water level in an estuary. Insufficient attention has been given to the ecological benefits. This paper aims to develop a model to allow valuation of one of the ecological benefits of managed realignment; the benefits to the fishing industry in the inshore waters near to the created salt marsh.

1.2 Salt marsh-Fishery Linkages

Society has reason to value the salt marsh areas of the intertidal zone of the UK. Its high primary productivity sustains an exceptional biodiversity especially in fish and molluscs (Laffaille, Feunteum & Lefeuvre, 2000). North American salt marshes are known to play a trophic and nursery role for many important fishery species (Mitsch & Gosselink, 2000). Comparatively, fish communities using European salt marshes have rarely been studied although these intertidal habitats play important roles for fish communities (Laffaille et al, 2000; Mathieson, Cattrijsse, Costa, Drake, Elliott, Gardner & Marchand, 2000; West & Zedler, 2000; Tupper & Able, 2000; Laffaille, Lefeuvre, Schricke & Feunteun, 2001).

The extent to which the composition of the fish community using the UK salt marshes has been studied has been limited by the difficult sampling methods required in collecting the data. In Europe, mean tide level borders the marsh. As a result, salt marshes and their intricate networks of creeks are flooded only during high spring tides. Fish therefore only invade this environment during short immersion periods (Laffaille *et al*, 2000). This potentially critical period where freshwater, brackish and marine species can feed and gain shelter is often overlooked. The nursery locality is usually described as being restricted to the mudflats of marine coastal waters such as estuaries and lagoons.

A conceptual model describing the structure and function of the salt marsh ecosystem is shown in figure 1 (from Mitsch & Gosselink, 2000). It is the flow of energy to juvenile fish and adult and predatory fish in the bottom right side of diagram that is of interest. The complexity of the other interactions in the ecosystem means that each discrete unit area of salt marsh is unique. Salt marsh clearly represents a significant resource in the life cycle of many commercially important species.

The invertebrate fauna of mudflats and salt marshes provide a substantial food source for wading birds and are thought to represent important stops for weight gain during migration (Little, 2000). Fish species classed as “residents” of estuaries (typically gobies) are found throughout the year in salt marshes. However, given the low frequency with which salt marshes are tidally inundated, other patterns of use by commercially important fish species are described. Elliott, O’Reilly & Taylor (1990), use the terms “migrant”, “marine straggler” and “marine opportunist” to describe the patterns of use of estuarine water by 36 fish species present in their survey of the Firth of Forth. Marine opportunists drift into estuaries as larvae from eggs spawned in coastal waters. When young, these species live just above the bottom (demersal zone) and take advantage of the rich benthic food sources (Little, 2000). Salt marshes and mudflats therefore contain a large number of 0 group fish (individuals of less than 1 year of age) using them as nursery grounds before emigrating to the open ocean as recruits for adult populations.

In possibly the most detailed ecological study of fish use of salt marsh in Europe, Laffaille *et al* (2001) describe the feeding ecology of sea bass in their first year of life (0-Group fish). Their findings from a study site in Mont Saint Michel Bay, France show that 0-group sea bass colonise the intertidal marsh creeks of the bay on the spring tides during flood and return to coastal waters during ebb. Their stays are of 1 to 2 hours in duration and the fish consume on average a minimum of 8% of their body weight, most having arrived with empty stomachs. This is clearly a significant resource for the juvenile fish and one that must be crucial to their survival to recruitment to the adult stock.

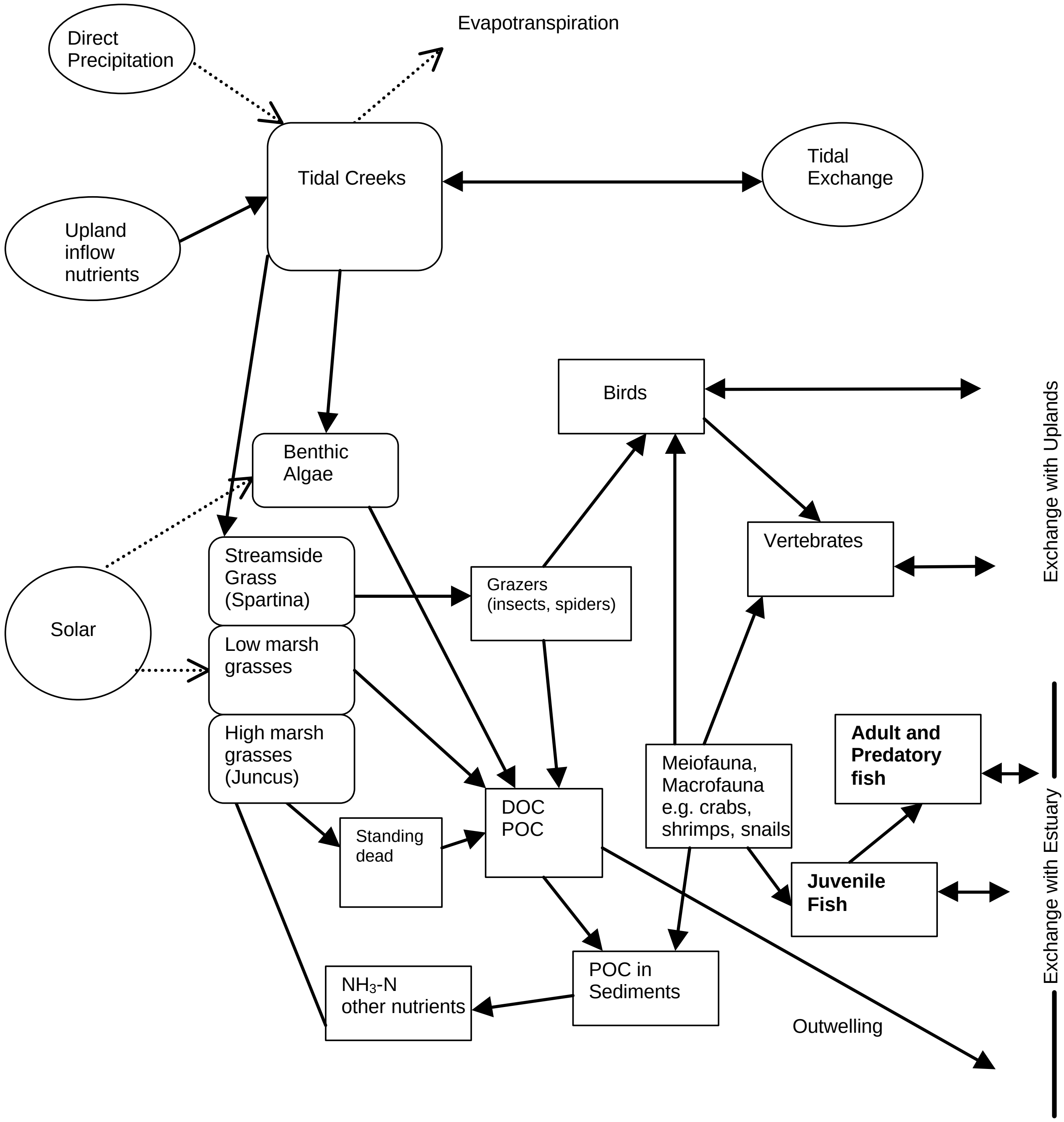

Figure 1 – Flow model of a tidal salt marsh (from Mitsch & Gosselink, 2000)

1.3 Economic Valuation of Salt marsh

Pearce & Turner (1990) describe the taxonomy of the quantifiable Total Economic Value (TEV) of an environmental resource. A classification in the context of wetlands is given by Barbier (2000) and is shown in fig. 2. The production function approach to valuation of indirect benefits to fisheries can be considered a partial valuation of TEV. This should be supplemented by valuation of the other outputs, benefits and values outlined below.

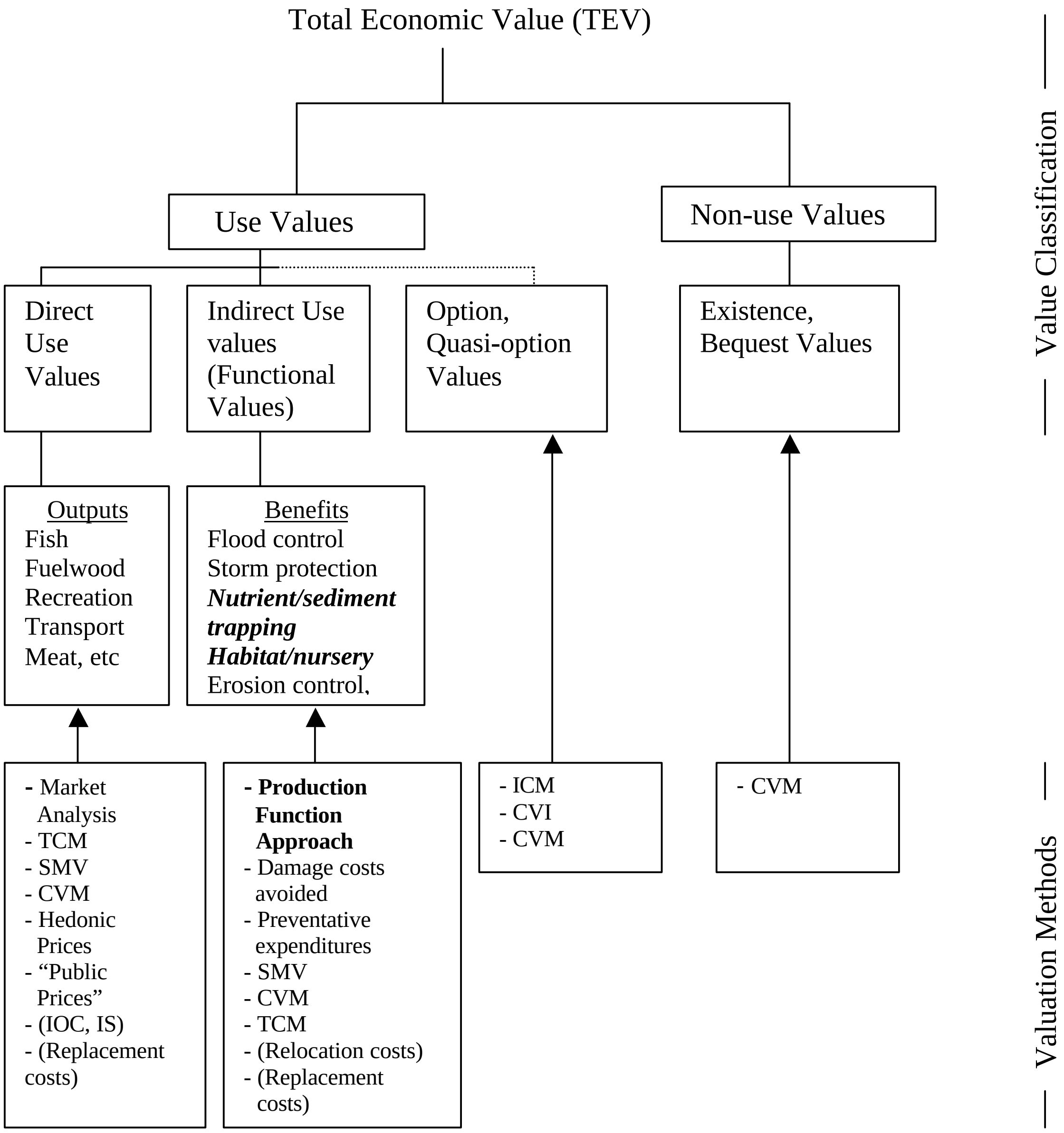

Figure 2 - Valuing wetland benefits (from Barbier, 2000)
Where,
ICM = Individual choice Models,
CVM = Contingent valuation method,
SMV = Surrogate market valuation,
IS = Indirect substitute approach
CVI = Conditional value of information models
TCM = Travel cost method
IOC = Indirect opportunity cost approach
(…) = Methods in to be used with care

The current project relates to the process of valuation of the benefits marked in bold type in the classifications in figures 1 and 2. The methodological approach can broadly be termed the “production function” method (after Barbier, 2000). The few applications in the literature have been to areas of mangroves in the tropics (Sathirathai, 1998; Barbier & Strand, 1998). However, there is definite validity to application to salt marsh habitat in temperate zones. Mudflats throughout the world are usually fringed at the landward side by wide reaches of salt-tolerant vegetation growing below the high-water mark; in the tropics these are mangroves and in the temperate zones these are salt marshes (Little, 2000).

The methodology relies on supply-side production or cost data. These concepts are easier for non-economists to understand and accept than valuation of “demand” for environmental goods; the conceptual basis for contingent valuation. Gosselink, Odum & Pope (1974, cited in Ellis & Fisher, 1987) were the first to attempt to establish a link between wetlands area and fisheries catch, presenting a crude 1:1 linkage. However, as Bell (1997) points out, this attributes all the economic value of the fishery to the wetland, ignoring all the other inputs to production.

Lynne, Conroy & Prochaska (1981) presented the first paper to attempt to isolate the contribution of an environmental input to production in this context. Lynne *et al* (1981) developed a bioeconomic model where human effort and wetlands were distinct inputs to the production of blue crab on Florida’s Gulf Coast. However, no measure of welfare change (resulting from changes in the area of wetland) was investigated.

Ellis & Fisher (1987) developed this work to approximate the production function of Lynne *et al* (1981) to a Cobb-Douglas form[1]. They then estimated changes in consumer and producer surplus associated with changes in wetland area. The Ellis-Fisher model is a static optimisation model of a fishery under a private property regime. The central tenet of such a model is the production function; an expression of outputs as a function of inputs into a productive process. By including salt marsh habitat as an input into the “production” of fish by fisheries around the coast of the

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[1] The Cobb-Douglas production function is the most popular mathematical form of production function that is assumed in agricultural economics.

UK, it is possible to estimate the elasticity of production of the salt marsh habitat using econometric techniques.

The elasticity of production for salt marsh, in a coastal fishery, is the proportionate change in fish supplied, divided by the proportionate change in salt marsh area. The larger the value of the elasticity, the more responsive fish supply is to changes in salt marsh area. For example, if the production elasticity is greater than 1, then a 1% change in salt marsh brings about more than a 1% change in production. Production is said to be *elastic* with respect to salt marsh area. If the elasticity of production is less than 1, the same change in salt marsh would result in a less than 1% change in quantity; production is said to be *inelastic* with respect to salt marsh. Elasticity of production of all inputs to production can be estimated by econometrics in this way.

Under the Ellis-Fisher model, an environmental improvement, such as the additional salt marsh area created by managed realignment, would result in a supply shift downward and to the right (see fig. 4 below). For optimally managed fisheries, the area between old and new supply curves indicates a welfare measure that is preferred in economic theory; a change in combined consumer and producer surplus. Firms are therefore profit maximising and the supply curve is based on marginal cost. As Freeman (1991) observes, the management regime affects the valuation output from the Ellis-Fisher model. Under open access, competition drives producer surplus (or rents) to zero and so the supply curve is average cost. Therefore changes in consumer surplus alone give the indirect-use value of the wetland (fig. 3).

Analysis of this kind is subject to criticism on the basis of the assumptions necessary for the model; it is a static partial equilibrium analysis. A recent paper has attempted a dynamic analysis, incorporating the production function into inter-temporal models of renewable resource harvesting (Barbier and Strand, 1998). Such an approach requires much greater data input but with greater predictive validity. However, the application of dynamic models to national studies of habitat valuation is not possible due to the level of resolution and volume of the data required.

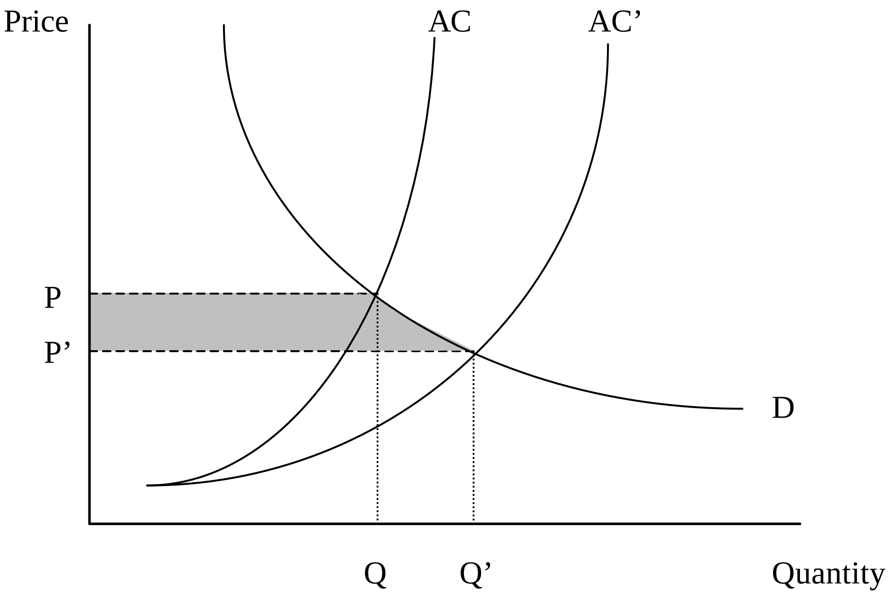

Fig. 3 - Supply shift and welfare change in an open-access fishery, as a result of an increase in salt marsh area. The average cost curve moves from AC to AC' and the change in consumer surplus is given by the shaded area.

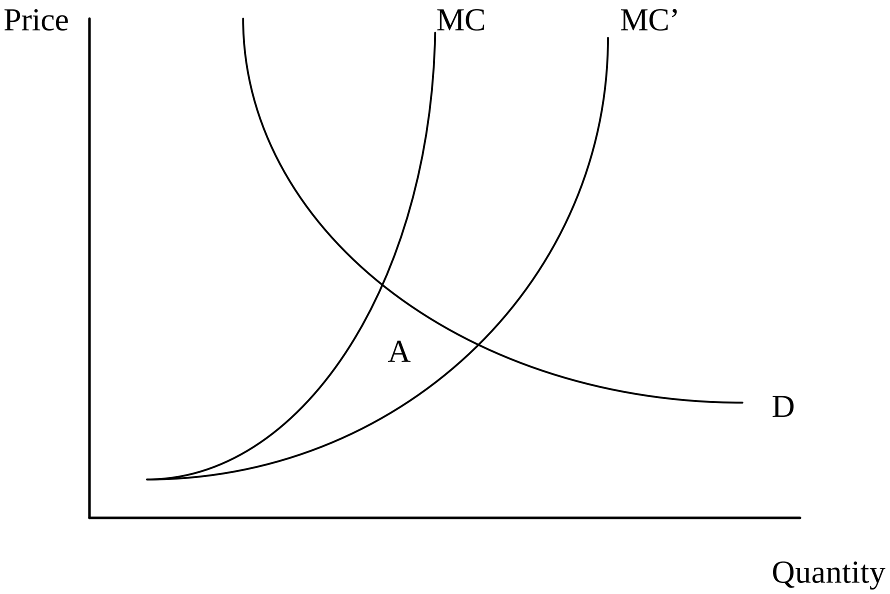

Fig. 4 - Supply shift and welfare change in an optimally managed fishery, as a result of an increase in salt marsh area. The combined producer and consumer surplus is given by the area A, bounded by the marginal cost curves and the demand curve.

Fig. 3 and 4 adapted from Sathirathai (1998)

1.4 Policy Context

The motivation for the valuation study is the need to better quantify the benefits of managed realignment. The approach adopted here is one of calculating a feasible and robust partial valuation for a unit area of salt marsh. There is a need for this valuation because the most influential pieces of evidence regarding the efficiency of a particular treasury funded project are the cost-benefit ratio and the Net Present Value (NPV). Both of these measures require ecosystem functions and services to be valued as fully as possible, to ensure the analysis is economic rather than merely financial.

The use of benefits transfer and other generic valuation measures is therefore advocated in the case of projects where habitat is to be created. The scientific uncertainty relating to the speed and pattern of colonisation of new growth and restored marshes is matched by inherent uncertainty regarding the specificity of benefits transfer valuations. Benefits transfer in the valuation of established wetlands should be approached with extreme caution. The need for site-specific study remains in this case. A combination of environmental economic and purely biological approaches would be necessary to inform the appraisal process.

Detailed biological surveys should ideally be carried out to assess the patterns of use of the habitat by commercially important and endangered species (see Laffaille *et al*, 2001, for an example). Certainly, the use of expert witness statements from estuarine and coastal biologists is strongly recommended. However, modelling exercises, such as the one included in this paper, can play an important role in quantifying the benefits of sustainable flood defence strategies that have indirect benefit to the fishing industry.

In terms of project appraisal, the time, expertise, and therefore money required to dynamically model each project is likely to be prohibitive. A dynamic model, estimated using primary and/or secondary data on fish stocks, growth rates, fishing effort, fisheries catch, technical efficiency and habitat area, represents the “ideal world” situation. In reality, these data requirements are prohibitive. Project appraisal practitioners are therefore likely to be reliant on the kind of static partial equilibrium model offered by Ellis & Fisher (1987). If managed realignment projects are to be appraised using CBA, then a transferable benefit function would be a useful tool for practitioners of project appraisal.

2. METHODS

2.1 The Ellis-Fisher-Freeman Model

The following section is the exposition of a technical economic model that underlies the welfare calculations that would be performed if significant elasticities of production can be established for salt marsh area and fishing effort via econometric estimation. It can be skipped by readers wanting a non-specialist understanding, although efforts are made to make it accessible to non-economists.

Sathirathai (1998) presents the model with clarity and it is often that author's notation and calculus presented here (adapted to the nature of the situation and type of data included).

For coastal fisheries, the optimisation problem faced by a price-taking firm is:

$$\max_{\bar{A}} \quad P * f(E, \bar{A}) - cE \qquad \text{Eq. 1}$$

where, P is Price; E is human effort (in days or hours fished); $\bar{A}$ is the wetland area in hectares which is considered exogenous in this problem; and c is the unit cost of effort.

X_s is quantity of fish caught (quantity supplied) which, it is assumed, depends on human effort and the area of wetland as represented by the Cobb-Douglas production function:

$$X_s = f(E, \bar{A}) = mE^a\bar{A}^b \qquad \text{Eq. 2}$$

where m, a and b are unknown positive constants, to be determined by econometric estimation.

The duality of the profit maximisation is the minimisation of the cost of effort (Sathirathai 1998). Therefore:

$$\min_E \quad L = cE + \lambda(X_s - mE^a\bar{A}^b) \qquad \text{Eq. 3}$$

Differentiating this expresssion, known as a Langrangean (L), with respect to the effort variable and the Langrange multiplier (the name for a notional variable introduced to help in finding constrained maxima and minima, denoted by λ), yields:

$$\frac{\delta L}{\delta E} = c - \lambda m\ \bar{A}^{b} a E^{a-1} = 0 \qquad \text{Eq. 4}$$

$$\frac{\delta L}{\delta \lambda} = X_s - mE^{a}\ \bar{A}^{b} = 0 \qquad \text{Eq. 5}$$

From equation 5, E can be solved for the cost function, $C(c, X_s, \bar{A})$:

$$E = \left[\frac{Xs}{m\bar{A}^{b}}\right]^{1/a} \qquad \text{Eq. 6}$$

which yields the cost function:

$$C\ (c, X, \bar{A}) = cm^{-1/a}\ X_s^{\ 1/a}\ \bar{A}^{b/a} \qquad \text{Eq. 7}$$

Ellis and Fisher (1987) assume that the fishery is under a private property rights regime. Therefore, price (P) is equal to marginal cost (MC). By differentiating the cost function with respect to output, the marginal cost can be expressed:

$$P = MC = \frac{\delta C}{\delta X} = (c/a)\ m^{-1/a}\ \bar{A}^{-b/a}\ X^{(1-a)} \qquad \text{Eq. 8}$$

However, Freeman (1991) has argued that in fact most fishery resources are open access in which rents are dissipated. In this situation, total revenue is equal to total cost, so price equals average cost (AC). The unit cost of effort, c, therefore has to be consistent with zero profit.

$$P = AC = C(c, X, \bar{A}) = cm^{-1/a}\ \bar{A}^{-b/a}\ X^{(1-a)/a} \qquad \text{Eq. 9}$$

A useful property of the Cobb-Douglas production function (Eq. 2) is that it is log-linear:

$$\ln X = \ln m + a \ln E + b \ln \bar{A} \qquad \text{Eq. 10}$$

Eq. 10 can therefore be used to estimate m, a, and b by econometrics, when data on X, E and Ā are known. It is these estimates of m, a and b that will determine the production function of UK fisheries, with salt marsh area as a component.

2.2 Data Collection

The spatial unit of analysis is the rectangle, set by the International Council for the Exploration of the Sea (ICES) as shown in figure 3. Fishermen record the number of days spent fishing, classified by the ICES rectangle in which they fish and the landings by species, by rectangle of capture.

2.2.1 Landings Data

Landings data were obtained via an extraction from the fisheries statistics databases of the Department for Environment, Food and Rural Affairs (DEFRA) and the Scottish Executive Rural Affairs Department (SERAD). Landing data by species, by ICES rectangle of capture for the years 1997-2001 are used for the analysis.

2.2.2 Effort Data

Effort data were also obtained from the SERAD and DEFRA databases for the years 1997-2001. The measure of effort is the least certain of the different types of data point. “Days fished” differs from “days at sea” and it is arguable as to which of these measures should be used for the analysis. Neither is perfect, particularly given the problems of boats visiting different rectangles on the same trip, differences in the length of time within one day spent in each rectangle by each boat and the efficiency of different gear types and boat sizes. The difficulty of measuring fishing effort absolutely is well known, and so the use of a proxy of days fished is an acceptable second-best approach for the purposes of the project.

2.2.3 Salt Marsh Data

Time-series data on the area of salt marsh for coastline bordering the ICES rectangles are not available. There is no annual reporting of salt marsh area in the same way there is for fisheries statistics. There is no variation, therefore, in the time dimension of the salt marsh data. However, there is a great deal of spatial heterogeneity in the data. The data on area coverage of salt marsh are gathered from the Salt Marsh Survey of Great Britain (Burd, 1989). This is the only national survey of intertidal habitat found in the literature.

The maps from Burd (1989) were superimposed with the ICES rectangle grids and the area of salt marsh for each coastal rectangle was calculated. However, it is hoped that spatial variability, combined with temporal variability in the effort variable and the dependent variable, will be sufficient to allow significant econometric estimation.

2.3 Multiple Regression Models

Consider equation 10 from the model outlined previously:

$$\ln X = \ln m + a \ln E + b \ln \bar{A}$$

This is the log-linear form of the Cobb-Douglas production function. The natural logarithm of both sides of the production function are taken so that the function is linear in the parameters (a and b). If a function is linear in the parameters (i.e. not in a form where the parameter being estimated states the power to which an explanatory variable is raised) then estimation by ordinary, pooled or generalised least squares is available.[3]

Multiple regression refers to the fact that there is more than one explanatory variable in the regression; in this case Effort (E) and salt marsh area ($\bar{A}$). X is the quantity supplied (landings for an ICES rectangle) and is the dependent variable in the multiple regression. In order to establish the values of m, a and b from equation 10, a series of data points are required where variation in the dependent variable can be determined by variation in the explanatory variables.

Cross-section econometrics is used where there is spatial heterogeneity in data points but no time dimension to the data (e.g. a number of zones for one year). Time-series econometrics is used where there is temporal variability in the data but no spatial dimension (e.g. one zone over a number of years). In the case of mixed temporal and spatial variation, as is the case here, panel econometric methods are used (see section 2.5 below).

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[3] Greene (2000) gives a good introduction to econometric analysis.

2.4 Data Coding

11 coastal fishing zones represent the cross-section variability in the data. These were established on the basis of the ICES rectangles that border the coastline on the seaward side. The zones were chosen on the basis that they are relatively independent and are likely to be biologically discrete to an extent, due to tidal exchange. The 11 zones are shown in table 1 below (see figure 5 for illustration of the areas in question).

Zone	Geographic Area	ICES Rectangles
1	Thames Estuary	F0 32
		F0 31
		F1 32
		F1 31
2	The Wash	F0 34
		F0 35
3	Mersey and Dee	E7 37
		E7 36
		E7 35
		E6 36
		E6 35
4	Ceredigion	E5 34
		E5 33
5	Firth of Forth	E6 41
		E7 41
		E7 40
6	Inverness	E6 44
		E5 44
7	Solway Firth	E6 38
		E7 38
		E5 38
8	Bristol Channel	E7 32
		E7 31
		E6 32
		E6 31
9	Shetland	E8 50
		E8 49
		E9 50
		E9 49
10	Argyle & Bute	E5 40
		E4 40
		E4 41
		E3 41
11	Humber	E9 36
		F0 36

Table 1 – The 11 fishing zones used in the analysis, their geographic areas and ICES rectangles.

Once the data had been coded for the 11 zones it became apparent that zones 8 and 9 did not support the same level of fisheries activity as the other zones and so were excluded from the regression analysis. There are therefore 5 years of data for 9 cross-section units, giving a balanced panel of 45 observations.

2.5 Panel Econometric Estimation

In econometrics, a panel data set is the combination of cross-section data and time-series data, forming a matrix of data points. Two panel data sets used in the estimation of the Cobb-Douglas coefficients ("all species and gears", and "shellfish species and gears") are shown in appendix 2 and 3 respectively. In the current study, the cross-section units are the 9 fishing zones. It is important to recognise that these zones are heterogeneous; they are naturally productive to different extents.

While it is impossible to address this heterogeneity completely, the two panel econometric methods detailed below allow individual zones to differ, while using data from all zones in the estimation of the production function.

2.5.1 Fixed Effects Model[4]

Fixed effects models assume that the differences across zones can be captured in differences in the intercept (m in equation 10 above). Each zone has its own intercept that is an unknown parameter to be estimated, the same as the effort and salt marsh parameters. For 9 zones, 8 dummy variables (D_i) are introduced so that the regression equation looks like;

$$\ln X_i = m_i + a \ln \bar{A} + b \ln E_i + D_i + e_i$$

Fixed effects models are employed when the differences between cross-sectional units are considered to be deterministic. This produces parametric shifts in the regression function, such that separate values for m are estimated for each unit.

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[4] The distinction between an "economic model" and an "econometric model" should be made clear. In the current study, the Ellis-Fisher-Freeman model is an economic model, ultimately describing a possible relationship between salt marsh area and human welfare, via a number of intermediate equations that are formulated by reference to economic theory.
An econometric model is simply the specification of the relationships between components of the data sets that are to be used in the estimation of the parameters of the economic model.

2.5.1 Random Effects Model

Random effects models view the individual unit specific constant terms as randomly distributed across cross-sectional units. This alternative model is useful if the individual units appearing in the sample are randomly chosen and taken to be representative of a larger population of firms. While the fishing zones are not randomly chosen, it is likely that significant regression results would be inferred to apply to the UK coast as a whole.

Given the uncertainty as to which econometric model is the most appropriate, results from both specifications are presented in section 3. Also presented are results from simply pooling time-series and cross-section data, in a pooled least squares estimation.

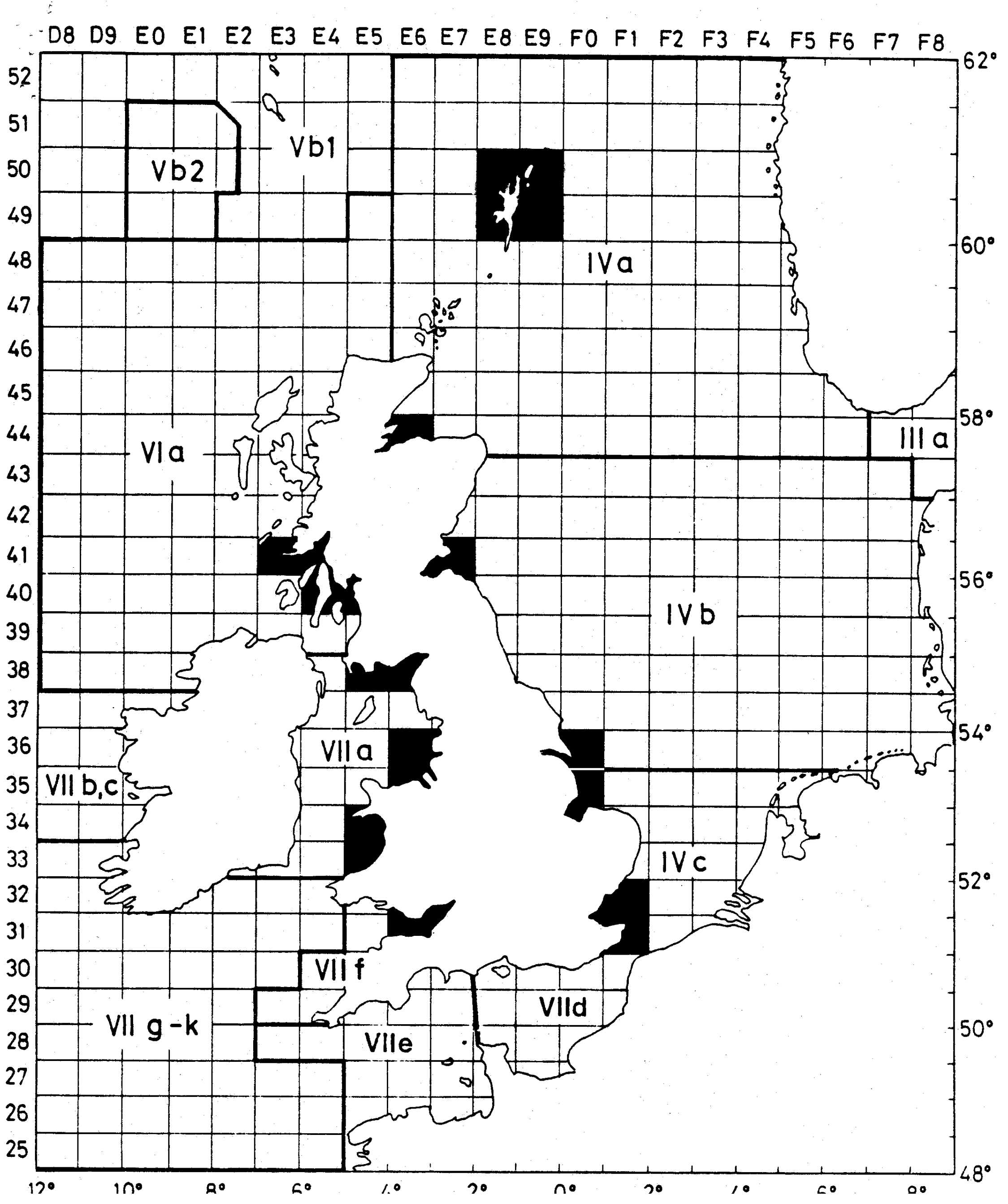

Figure 5 - ICES rectangles surrounding the coastline of the UK with the discrete fishing zones marked by shading

2.6 Emergent Methodological Issues

In carrying out the work, two main points of criticism emerged relating to the Ellis-Fisher-Freeman model and its subsequent econometric estimation using secondary data sets.

2.6.1 Jointness

Fisheries management and stock statistics are set at the level of the individual species. However, with very few exceptions species are targeted jointly – fishers can't target individual species. This has implications for the assessing the appropriate Total Allowable Catches (TACs) for the species but also for the estimation of production elasticities such as those that form the output from econometric analysis. The relationship between the effort variable and the catch becomes less explicit.

Frere & Pascoe (2002) show preliminary results for the estimation of the degree of jointness between species for the major English Channel fisheries species. Their findings suggest that most species are targeted jointly, rather than as separate products. The implication for production function estimation is that a unit of fishing effort produces multiple outputs. Clearly in the case of trawlers this is intuitively obviously – a valuable or abundant species might be targeted particularly but that many other species are caught in the process. These might be discarded at sea and would not appear in the landings statistics. If the by-catch is sufficiently large for a given species however, it will be sorted and sold.

The most species-specific capture methods are passive gears, where the target species move into the gear once set. These are particularly common in lobster fishing (pots) although gill nets are still sometimes set to passively catch demersal fish with some specificity. The active method of purse seining, where a shoal of pelagic fish is encircled by a net and drawn close like a string purse, is also likely to be fairly species specific. The potential for other mid-water fish and sea creatures to become trapped in this process is obvious. Opposition to purse seining led to consumer-placating marketing strategies such as "dolphin friendly tuna". Pelagic fish (e.g. mackerel, herring) do not rely on the very shallow waters for any part of their lifecycle. Therefore, no significant relationship between intertidal marsh area and the catches of these species is expected. The demersal fisheries, for which one would expect the greatest contribution from intertidal areas, use the gear types that are the least species-specific. This means that the explanatory power of the salt marsh variable in the regression will be either underestimated or not statistically significant.

The specificity of the shellfish gears is the greatest. If regression in aggregate is insignificant due to noise effects, shellfish provide the subset most likely species subset to counter the problems of jointness.

2.6.2 Stock Measures

In using a fisheries panel data set, the time dimension demands that previous year's catch be considered in the analysis of subsequent years. This is because the level of harvesting in a given year *(t)* affects the level of harvesting in the following year *(t+1)*. This is because fish stocks can be considered as a renewable resource. Growth of a stock over year t occurs when the harvesting from that stock is less than growth of the stock (determined species reproduction and biological growth rates). The harvesting in year *t+1* is thus a function of the stock and harvesting from year *t*.

This introduces a clause in the model that is assumed away in the analysis of Ellis-Fisher-Freeman. Their model is strong on the user-friendly nature of the welfare calculations (and therefore monetary valuation) arising from changes in the salt marsh area. While this appeals to practitioners of project appraisal, the biological basis for such a static analysis is limited. In a dynamic approach, the salt marsh "support function" for fisheries is included in the intertemporal bioeconomic harvesting problem (Barbier, 2000). In this respect, the salt marsh input to fisheries production will be modelled as part of the growth function of the fish stock. Pragmatism in terms of what can be achieved in a short period of time with secondary data sets that are gathered for another purpose, has forced the current study away from dynamic modelling. However, dynamic models should be considered best practice if the resources and data are available.

3. RESULTS

3.1 All Species and Gear Types[5]

Variable	Coefficient	Std. Error	t-Statistic	Prob.
M	4.673494	0.968028	4.827851	0.0000
LN EFFORT	0.538904	0.161658	3.333605	0.0018
LN SALTMARSH	-0.111361	0.156108	-0.713357	0.4796

R-squared	0.276105	Mean dependent var	8.038012
Adjusted R-squared	0.241634	S.D. dependent var	0.939950
S.E. of regression	0.818548	Sum squared resid	28.14086
F-statistic	8.009733	Durbin-Watson stat	0.336596
Prob(F-statistic)	0.001130		

Table 2a – Common Intercept Estimation (pooled least squares)

Variable	Coefficient	Std. Error	t-Statistic	Prob.
LN EFFORT	0.786459	0.302433	2.600438	0.0137
LN SALTMARSH	0.229167	1.99E+14	1.15E-15	1.0000
Fixed Effects				
_ONE—M	-0.099719			
_TWO—M	-0.380938			
_THRE—M	-0.136335			
_FOU—M	-1.433622			
_FIV—M	1.669640			
_SIX—M	0.442544			
_SEV—M	0.014111			
_TEN—M	1.604598			
_ELE—M	-0.213428			

R-squared	0.863718	Mean dependent var	8.038012
Adjusted R-squared	0.823635	S.D. dependent var	0.939950
S.E. of regression	0.394739	Sum squared resid	5.297848
F-statistic	215.4831	Durbin-Watson stat	1.737303
Prob(F-statistic)	0.000000		

Table 2b – Fixed Effects Estimation (pooled least squares)

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[5] In reading these results tables, two important figures should be considered for each variable. “Coefficient” is the estimated production elasticity in the case of effort and saltmarsh, and the value of the intercept in the case of ‘m’. Values of “p” (Prob. in the right-most column) should be less than 0.05, the estimates of the coefficients to be considered statistically significant.

Variable	Coefficient	Std. Error	t-Statistic	Prob.
M	4.261046	2.105808	2.023473	0.0494
LN EFFORT	0.689344	0.235789	2.923561	0.0056
LN SALTMARSH	-0.212039	0.305917	-0.693127	0.4920
Random Effects				
_ONE—M	0.398118			
_TWO—M	-0.011357			
_THRE—M	0.352769			
_FOU—M	-1.700632			
_FIV—M	0.510922			
_SIX—M	-0.047903			
_SEV—M	0.156348			
_TEN—M	0.796519			
_ELE—M	-0.454784			
GLS Transformed Regression				
R-squared	0.838540	Mean dependent var		8.038012
Adjusted R-squared	0.830851	S.D. dependent var		0.939950
S.E. of regression	0.386579	Sum squared resid		6.276634
Durbin-Watson stat	1.477780			
Unweighted Statistics including Random Effects				
R-squared	0.862286	Mean dependent var		8.038012
Adjusted R-squared	0.855729	S.D. dependent var		0.939950
S.E. of regression	0.357022	Sum squared resid		5.353511
Durbin-Watson stat	1.732598			

Table 2c – Random Effects Estimation (generalised least squares – variance components)

The results from all three econometric specifications show no significant production elasticity for salt marsh, for the all species data set. Significant, inelastic (in the range 0.5 to 0.8 but all less than 1) coefficients for the effort variable, are estimated for all econometric specifications, as would be expected by production economic theory; increasing fishing effort by 1% increases landings but by less than 1%.

3.2 Shellfish Species and Gear Types

Variable	Coefficient	Std. Error	t-Statistic	Prob.
M	5.919459	0.987466	5.994597	0.0000
LN EFFORT	0.043677	0.064833	0.673681	0.5042
LN SALTMARSH	0.202016	0.129985	1.554140	0.1277
R-squared	0.078094	Mean dependent var		7.741884
Adjusted R-squared	0.034194	S.D. dependent var		0.935070
S.E. of regression	0.918944	Sum squared resid		35.46724
F-statistic	1.778903	Durbin-Watson stat		0.730776
Prob(F-statistic)	0.181308			

Table 3a – Common Intercept Estimation (pooled least squares)

Variable	Coefficient	Std. Error	t-Statistic	Prob.
LN EFFORT	-0.022801	0.041272	-0.552465	0.5842
LN SALTMARSH	60355600	68154083	0.885576	0.3821
Fixed Effects				
_ONE—M	-5.27E+08			
_TWO—M	-5.15E+08			
_THR—M	-5.45E+08			
_FOU—M	-4.31E+08			
_FIV—M	-3.50E+08			
_SIX—M	-4.26E+08			
_SEV—M	-5.05E+08			
_TEN—M	-3.75E+08			
_ELE—M	-4.38E+08			
R-squared	0.749594	Mean dependent var		7.741884
Adjusted R-squared	0.675945	S.D. dependent var		0.935070
S.E. of regression	0.532296	Sum squared resid		9.633535
F-statistic	101.7795	Durbin-Watson stat		2.392752
Prob(F-statistic)	0.000000			

Table 3b – Fixed Effects Estimation (pooled least squares)

Variable	Coefficient	Std. Error	t-Statistic	Prob.
M	6.072079	1.827854	3.321971	0.0019
LN EFFORT	-0.004829	0.039574	-0.122017	0.9035
LN SALTMARSH	0.224870	0.239200	0.940093	0.3525
Random Effects				
_ONE—M	0.965213			
_TWO—M	0.268543			
_THR—M	-0.341717			
_FOU—M	-1.196463			
_FIV—M	-0.104211			
_SIX—M	-0.536180			
_SEV—M	-0.476999			
_TEN—M	0.986131			
_ELE—M	0.435683			
GLS Transformed Regression				
R-squared	0.676997	Mean dependent var		7.741884
Adjusted R-squared	0.661616	S.D. dependent var		0.935070
S.E. of regression	0.543938	Sum squared resid		12.42646
Durbin-Watson stat	1.913304			
Unweighted Statistics including Random Effects				
R-squared	0.737105	Mean dependent var		7.741884
Adjusted R-squared	0.724587	S.D. dependent var		0.935070
S.E. of regression	0.490723	Sum squared resid		10.11399
Durbin-Watson stat	2.350763			

Table 3c – Random Effects Estimation (generalised least squares – variance components)

It can been seen that there are no longer even significant coefficients for the effort variable, and definitely not the positive inelastic coefficient for the salt marsh variable that would be used in subsequent welfare calculations. The loss of significance of the effort coefficient relates to the fact that an additional source of error is introduced when isolating the shellfish gear types and species from the data sets on all gears and species. Sources of error and non-significance in these econometric results, and their implications, are considered in the following section and should be considered alongside the problem of "jointness" that were outlined in section 2.6.1.

4. ANALYSIS AND DISCUSSION

4.1 Sources of Error

Sources of error relate to inaccuracies in compiling the data sets.

4.1.1 Accuracy of the Effort Data

The data for the under 10m vessel fleet is of very low reliability. There are no legal requirements for vessels to record the ICES rectangle in which they fish, unlike the >10m fleet. Clearly the smaller boats are those most likely to be fishing in waters closest to the coast and are therefore most important in determining the true fishing effort in these areas.

4.1.2 Aggregation of Inputs and Outputs

Fishing gears are not species specific. Fishing is therefore characteristic of a multi-output process. In 3.1, all species and gear types are included in the estimation. To give an illustrative example, it follows that a tonne of lobster is equivalent to a tonne of cod, and that a day’s lobster fishing is equivalent to a day spent trawling. This is clearly problematic in terms of the number of individuals that comprise a tonne of the species in question, the different effects of salt marsh on the species in question, as well as the relative efficiencies of the different gear types. In 3.2, only shellfish species are used to calculate the dependent variable. Correspondingly, only the fishing gears likely to target shellfish are used for the regression (species and gear types are shown in Appendix 1).

4.2 Reasons for Non-significance

Non-significance relates to inability to fit a regression line to the data that is consistent with the principles of the scientific method (e.g. the use of P values to communicate confidence in the clauses in the regression). The most likely reason for non-significance in the regression is that explanatory variables are missing from the function. In the “production of fish by the sea”, missing variables might typically include sea temperature, water quality, the degree to which habitat is sheltered from wave action, boat power or capacity or fishing gear efficiency. Altering the specification of the econometric model with regard to these potentially missing variables is not a difficult task. The problem is obtaining the data for each ICES rectangles for each of these potentially explanatory variables.

Simply increasing the sample size is increases the probability that the estimation of production elasticities is statistically significant. However, in the current study, this could only be done by including data from a larger number of years or by making the zones smaller and thereby increasing the number of cross-sectional units. The possibility for increasing the number of years in the time-series is limited by the fact that saltmarsh data isn’t available for more than one year. Equally, the size of the fishing zones could not feasibly be reduced without compromising the ecological credibility of the study (which is already fragile); some kind of link between juvenile feeding in salt marsh and adult capture in the surrounding water, has to be ensured.

4.3. Implications of the Findings

It is possible to conclude that the Ellis-Fisher-Freeman model cannot be applied at the national level. It is the missing problem of missing variables that is likely to be the binding constraint to its success. The Cobb-Douglas production function that is central to the Ellis-Fisher-Freeman model is simplistic for purposes of analytical convenience. This simplicity was necessary for the current study because a limited number of secondary sources were used for the estimation of production function elasticities.

In the case of Sathirathai (1998), a Cobb-Douglas form was sufficient to estimate a significant elasticity of production for mangrove area in a localised coastal fishery in Thailand. However, these data were for localised areas in which biological dependence of the fishery species on the habitat could be strongly inferred. In Sathirathai’s study, the loss of mangrove area was proceeding at a pace and there was temporal variability in the habitat data. Sathirathai’s study generated effort data by direct observation, rather than relying on secondary data sources, such as fishery statistics. In general, the links between fishing catch, effort and habitat area are much more certain.

The policy implications of successfully applying the principles in cross-section for the whole of the UK, in order to calculate a transferable benefit function, would be tremendous. The resulting elasticity of production for salt marsh would allow partial valuation of any salt marsh habitat, in terms of the benefit to fisheries, by means of benefit function transfer (Loomis, 1992). For any methodology, there is always a trade-off between accuracy and the possibility of wider use. In scaling up to the national level, the probability that the model will successfully capture the biological dependence of the fisheries catch on salt marsh habitat is greatly reduced. The effect of the “noise” from other, potentially unquantifiable variables, becomes overpowering.

These findings should be considered an update on the findings from Stevenson (2001). In that study, benefits to the shellfisheries of Scotland were found to be approximately £750 per hectare per year with a production elasticity for the salt marsh variable of 0.298. In that study, a significant production function relationship was established but the assumptions regarding the behaviour of the fishing fleet were much stronger than those made here. Therefore, consistent with the practice of relaxing assumptions where possible, the inconclusive findings from the current paper should be considered the *status quo*.

5. CONCLUSIONS

There are two possible conclusions that could be drawn from the non-significance of the salt marsh variable in the regression.

1) The Ellis-Fisher-Freeman model requires re-specification with other functional forms and additional explanatory variables but with improvements, this kind of model could yield useful results at the national level.
2) The simplifying assumptions in the Ellis-Fisher-Freeman model mean that the model is unlikely to be useful at the national level.

I think that conclusion (2) is the correct one. While the relatively small sample in the current study means that conclusion (1) cannot be rejected outright, the results suggest that other methodological approaches should be favoured. A recent review of the potential production function approaches that could be applied to intertidal habitat valuation is given by Barbier (2000). While Ellis-Fisher-Freeman is attractive due to the low number of data sets required, it can considered an oversimplification of a complex, dynamic system of interactions.

This is not to say that the results from the study are not useful. The current study highlights another reason why the process of benefit transfer, and the use of generic valuations to piece together TEV, should be approached with great caution. The nature of coastal ecosystems is one of great complexity. It doesn’t help to pretend that there is a simple “catch-all” methodology; there isn’t. The debate as to whether or not a framework of TEV to guide project appraisal is correct, is outside the scope of this study. However, in the case of generating transferable functions or per hectare values for salt marsh benefits to fisheries, the underlying models that generate such values must be as realistic as possible.

Clearly the use of static optimisation models, such as that of Ellis-Fisher-Freeman, is a more simplified approach to that of dynamic models. Whether large-scale dynamic bioeconomic models can be developed that can be used to inform appraisal at more local levels, will determine whether or not the benefits to fisheries can eventually be transferred for a generic cost-benefit analysis regime.

6. RECOMMENDATIONS FOR RESEARCH

If positive progress on the issue of valuing the fisheries benefits component of Total Economic Value (TEV) of salt marsh creation projects is to be made, a research project should be undertaken with the following suggested as guidance.

1. An interdisciplinary research team should be commissioned, with specialists in fisheries ecology, production economics and environmental economics.

2. The team should focus on wider issues than simply the methodological and data problems raised in the current study.

3. Static and dynamic modelling approaches should be assessed for the appropriate trade-off between ecological and production realism and the feasibility of obtaining the necessary data.

4. In the short to medium term, while production function modelling in environmental economics is still in its infancy, practitioners of project appraisal for flood defence should be well briefed on the benefits of salt marshes to fisheries. For large projects, this process should be informed by biological surveys (e.g. using the methods in Lafaille *et al* (2000).

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8. APPENDICES

Appendix 1

Species Used in 3.2

BROWN SHRIMPS
CLAMS (V.DECUSSATA)
COCKLES
CRABS (C.P.MIXED SEXES)
CUTTLEFISH
ENGLISH PRAWNS
LOBSTERS
NATIVE OYSTERS
MUSSELS
NEPHROPS
OCTOPUS
PACIFIC OYSTERS
PERIWINKLES
QUEEN SCALLOPS
SCALLOPS
SQUID
WHELKS

Bottom Gears Used in 3.2

BEAM TRAWL
BOTTOM PAIR TRAWL
HAND LINES (INC GURDY)
OTHER OR MIXED POTS
POWER DREDGE
TOP OPENING POTS
TWIN OTTER TRAWL
UNSPECIFIED DREDGE
UNSPECIFIED OTTER TRAWL
DRIFT NET
HAND DREDGE
PARLOUR POTS
TOP OPENING POTS
SUBMERGED PICKING
SUCTION DREDGE
LIGHT OTTER TRAWL
FYKE NET
TRIPLE OTTER TRAWL

Appendix 2

All Species and Gear Types

Zone	Year	ln landings	ln effort	ln saltmarsh
_ONE	1997	9.763506	9.809067	8.72585641
_ONE	1998	9.0624409	9.67376	8.72585641
_ONE	1999	8.9124546	9.553788	8.72585641
_ONE	2000	9.6847027	9.371949	8.72585641
_ONE	2001	9.5738528	9.269929	8.72585641
_TWO	1997	7.584642	8.604105	8.53927948
_TWO	1998	8.011935	8.749732	8.53927948
_TWO	1999	9.1735574	9.192278	8.53927948
_TWO	2000	9.0022444	9.004177	8.53927948
_TWO	2001	9.0334463	8.858795	8.53927948
_THRE	1997	8.1104209	8.221748	9.02765475
_THRE	1998	8.3414686	8.14584	9.02765475
_THRE	1999	7.9326066	8.084562	9.02765475
_THRE	2000	7.8705757	7.924072	9.02765475
_THRE	2001	9.0806169	7.896925	9.02765475
_FOU	1997	6.2855958	7.891331	7.14015189
_FOU	1998	6.3951631	7.54539	7.14015189
_FOU	1999	6.073701	7.674153	7.14015189
_FOU	2000	7.2390635	8.253228	7.14015189
_FOU	2001	6.2308259	8.321422	7.14015189
_FIV	1997	8.3247447	6.173786	5.79176283
_FIV	1998	7.6920207	5.971262	5.79176283
_FIV	1999	7.495903	5.921578	5.79176283
_FIV	2000	7.4193158	5.771441	5.79176283
_FIV	2001	7.087288	5.451038	5.79176283
_SIX	1997	7.8737786	6.958448	7.05717478
_SIX	1998	7.3017556	6.514713	7.05717478
_SIX	1999	7.2307905	6.728629	7.05717478
_SIX	2000	7.2010933	6.740519	7.05717478
_SIX	2001	7.2083175	6.774224	7.05717478
_SEV	1997	7.5328934	7.695758	8.37130925
_SEV	1998	7.8176697	7.349231	8.37130925
_SEV	1999	8.2536846	7.448916	8.37130925
_SEV	2000	7.4363503	7.211557	8.37130925
_SEV	2001	7.6346264	7.184629	8.37130925
_TEN	1997	9.1792469	7.80751	6.21704513
_TEN	1998	9.1051797	7.624619	6.21704513
_TEN	1999	9.2678533	7.367709	6.21704513
_TEN	2000	8.9809788	7.845024	6.21704513
_TEN	2001	8.7442591	7.667158	6.21704513
_ELE	1997	7.9619133	8.337109	7.25785566
_ELE	1998	7.6810985	8.151622	7.25785566
_ELE	1999	7.7584304	8.064636	7.25785566
_ELE	2000	7.9883292	8.223091	7.25785566
_ELE	2001	8.170195	8.307459	7.25785566

Appendix 3

Shellfish Species and Bottom Gear Types Data Set

Zone	Year	ln landings	ln effort	ln saltmarsh
_ONE	1997	9.423336	8.931684	8.72585641
_ONE	1998	9.10894	8.93945	8.72585641
_ONE	1999	8.5484102	2.757105	8.72585641
_ONE	2000	8.722449	9.241451	8.72585641
_ONE	2001	9.531169	9.372204	8.72585641
_TWO	1997	9.032547	8.856376	8.53927948
_TWO	1998	8.997626	8.99206	8.53927948
_TWO	1999	8.0759165	2.776152	8.53927948
_TWO	2000	7.947006	8.671458	8.53927948
_TWO	2001	7.21489	8.546752	8.53927948
_THR	1997	8.91635	7.758761	9.02765475
_THR	1998	6.747524	7.766417	9.02765475
_THR	1999	7.8909148	2.580486	9.02765475
_THR	2000	7.865983	8.090709	9.02765475
_THR	2001	7.029456	8.073403	9.02765475
_FOU	1997	6.093005	8.143517	7.14015189
_FOU	1998	7.134621	8.088255	7.14015189
_FOU	1999	6.133659	2.580013	7.14015189
_FOU	2000	6.164018	7.452402	7.14015189
_FOU	2001	6.063397	7.887959	7.14015189
_FIV	1997	7.013645	5.525453	5.79176283
_FIV	1998	7.392068	6.257668	5.79176283
_FIV	1999	6.9056518	2.656879	5.79176283
_FIV	2000	7.449741	5.913503	5.79176283
_FIV	2001	7.405546	6.120297	5.79176283
_SIX	1997	6.860243	6.706862	7.05717478
_SIX	1998	6.900881	6.6995	7.05717478
_SIX	1999	7.6679368	2.748626	7.05717478
_SIX	2000	6.555626	6.472346	7.05717478
_SIX	2001	7.194567	6.898715	7.05717478
_SEV	1997	7.283369	7.104965	8.37130925
_SEV	1998	7.191441	7.145196	8.37130925
_SEV	1999	7.714102	2.706752	8.37130925
_SEV	2000	7.534632	7.307202	8.37130925
_SEV	2001	7.24984	7.664347	8.37130925
_TEN	1997	8.503308	7.463363	6.21704513
_TEN	1998	8.47773	7.712444	6.21704513
_TEN	1999	8.2201926	2.716002	6.21704513
_TEN	2000	8.694747	7.434257	6.21704513
_TEN	2001	8.7635	7.710653	6.21704513
_ELE	1997	8.161823	8.299286	7.25785566
_ELE	1998	7.976818	8.211754	7.25785566
_ELE	1999	9.2423468	2.859243	7.25785566
_ELE	2000	7.521917	8.032685	7.25785566
_ELE	2001	7.861893	8.217978	7.25785566