

SPECIAL ISSUE

THE VALUES OF WETLANDS: LANDSCAPE AND INSTITUTIONAL
PERSPECTIVES

Valuing the environment as input: review of applications to
mangrove-fishery linkages

Edward B. Barbier *

*Director, Centre for Environment and Development Economics, Environment Department, University of York, Heslington,
York YO1 5DD, UK*

Abstract

The following paper reviews recent developments in the methodology for valuing the role of wetlands in supporting economic activity. The main focus will be on mangroves serving as a breeding ground and nursery habitat in support of coastal and marine fisheries. As this particular ecological function of a mangrove system means that it is effectively an unpriced 'environmental' input into fisheries, then it is possible to value this contribution through applying the production function approach. The first half of the paper overviews the procedure for valuing the environment as an input, applied to the case of a wetland supporting a fishery. Both the 'static' Ellis–Fisher–Freeman approach and the 'dynamic' approach developed by Barbier and Strand, incorporating the intertemporal bioeconomic fishing problem, are reviewed. The second half of the paper discusses briefly two recent case studies of mangrove-fishery valuation. An application in South Thailand, which is based on the static Ellis–Fisher–Freeman model, and an application in Campeche, Mexico, which is based on the dynamic approach. © 2000 Elsevier Science B.V. All rights reserved.

Keywords: Ecological functions; Environmental valuation; Fisheries; Habitat-fishery linkages; Mangroves

1. Introduction

The following paper overviews the general methodology for valuing mangrove-fishery linkages that can be applied to a variety of mangrove and coastal wetland systems found around the world. This approach has been used to assess the

economic value of coastal wetland habitats in support of marine fisheries and other ecological functions, such as determining the value of marshlands as habitat for Gulf Coast fisheries in the southern United States (Lynne et al., 1981; Ellis and Fisher, 1987; Farber and Costanza, 1987; Bell, 1989; Freeman, 1991; Bell, 1997) and the value of mangroves for coastal and marine fisheries in Thailand (Sathirathai, 1997) and Mexico (Barbier and Strand, 1998). This approach is consistent with other related studies attempting to

* Tel.: + 44-1904-434060; fax: + 44-1904-432998.

E-mail address: eb9@york.ac.uk (E.B. Barbier).

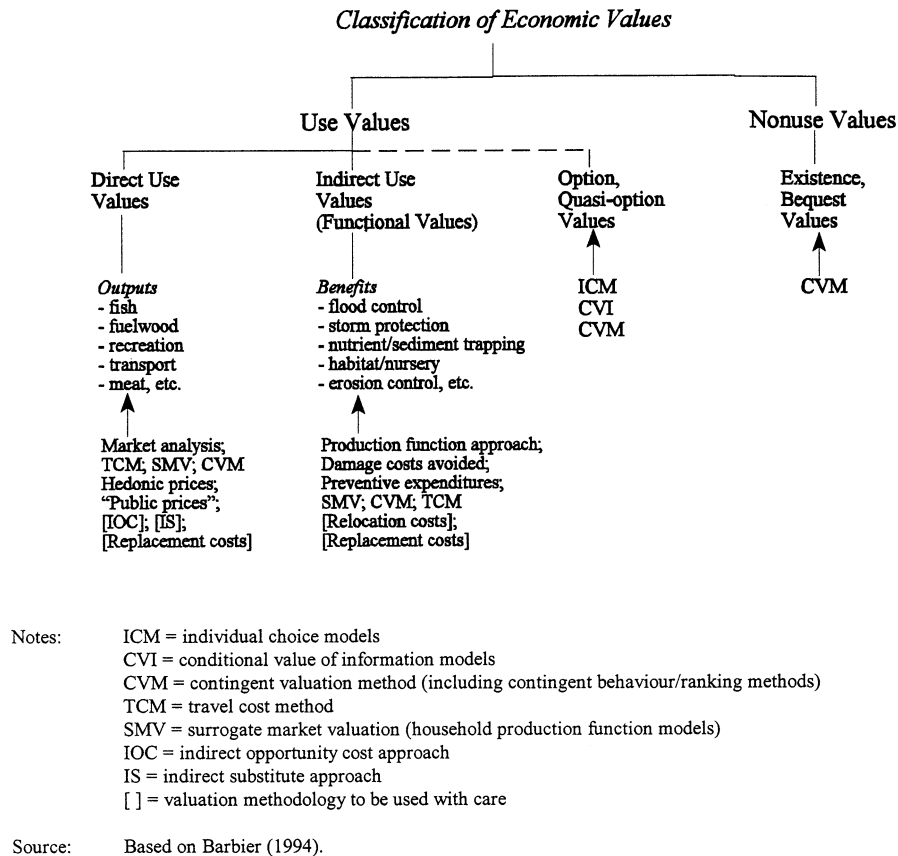


Fig. 1. Valuing wetland benefits.

analyze habitat-fishery problems more generally, such as analyzing the competition between mangroves and shrimp aquaculture in Ecuador (Parks and Bonifaz, 1994), determining the value of a multiple-use mangrove system under different management options in Bintuni Bay, Irian Jaya, Indonesia (Ruitenbeek, 1994), and examining general coastal system trade-offs, such as the effects of development and/or pollution on habitat-fishery linkages (Kahn and Kemp, 1985; Knowler et al., 1997; Strand and Barbier, 1997; Strand and Bockstael, 1990; Swallow, 1990; Swallow, 1994).

Natural wetlands, including mangroves, provide many important functions for humankind, which can be grouped in terms of direct use, indirect use and non-use values. Fig. 1 summarizes the standard techniques available for assessing the various economic values of wetlands.¹

Here, the concern is mainly with indirect use values, i.e. the indirect support and protection provided to economic activity and property by a wetland's natural 'services', or regulatory ecological functions. The ecological function of particular interest is the role of a mangrove or coastal estuarine wetland system in serving as a breeding ground or nursery for off-shore fisheries.

The main technique for valuing this ecological function of a wetland has been called, variously, the production function approach, valuing the environment as input and the value of changes in productivity approach (Freeman, 1991; Mäler, 1991; Barbier, 1994). The basic assumption of this

¹ For a guide to economic valuation of wetlands, see Barbier et al. (1997).

approach is that, because the wetland serves as a breeding ground or nursery for the fishery, this function can be treated as an additional environmental ‘input’ into the fishery. In static approaches, the welfare contribution of this input is determined through producer and consumer surplus measures of changes in the market equilibrium for harvested fish. In dynamic approaches, the wetland support function is included in the intertemporal bioeconomic harvesting problem, usually as part of the growth function of the fish stock, and any welfare impacts of a change in this function can be determined in terms of changes in the long-run equilibrium conditions of the fishery or in the harvesting path to this equilibrium.

The following section reviews both the static and dynamic production function approaches and their suggested applications to the wetland-fishery valuation problem. Two recent case studies of valuing mangrove-fishery linkages are then reviewed. One applies the static methodology in Southern Thailand (Sathirathai, 1997), and the other applies the dynamic model in Campeche, Mexico (Barbier and Strand, 1998).

2. The production function approach

When a wetland is being indirectly used, in the sense that the ecological functions of the wetland are effectively supporting or protecting economic activity, then the value of these functions is essentially nonmarketed. However, economists have demonstrated that it is possible to value such nonmarketed environmental services through the use of surrogate market valuation, which essentially uses information about a marketed good to infer the value of a related nonmarketed good. Travel cost methods, recreational demand analysis, hedonic pricing and averting behaviour models are all examples of surrogate market valuation that attempt to estimate the derived demand by households for environmental quality.

The following section describes another type of surrogate market valuation that is particularly useful for the valuation of nonmarketed values associated with biological resources and ecosys-

tems, such as coastal wetlands and mangroves, that protect or support economic activity, in particular coastal and marine fisheries. This is the production function approach.²

The general approach consists of a two-step procedure. First, the physical effects of changes in a biological resource or ecological function on an economic activity are determined. Second, the impact of these environmental changes is valued in terms of the corresponding change in the marketed output of the corresponding activity. In other words, the biological resource or ecological function is treated as an ‘input’ into the economic activity, and like any other input, its value can be equated with its impact on the productivity of any marketed output.

More formally, if Q is the marketed output of an economic activity, then Q can be considered to be a function of a range of inputs:

$$Q = F(X_1 \dots X_k, S) \quad (1)$$

For example, the ecological function of particular interest is the role of mangroves in supporting off-shore fisheries through serving both as a spawning ground and a nursery for fry. The area of mangroves in a coastal region, S , may therefore have a direct influence on the catch of mangrove-dependent species, Q , which is independent from the standard inputs of a commercial fishery,

² The production function approach discussed here is related to the household production function approach, which is a more appropriate term for those surrogate market valuation techniques based on the derived demand by households for environmental quality. That is, by explicitly incorporating non-marketed environmental functions in the modelling of individuals’ preferences, household expenditures on private goods can be related to the derived demand for environmental functions (Bockstael and McConnell, 1981; Freeman, 1993; Smith, 1991). Some well-known techniques in applied environmental economics — such as travel cost, recreation demand, hedonic pricing and averting behaviour models — are based on the household production function approach. The dose-response technique is also related to the production function and household production function approaches; however, dose-response models are generally used to relate environmental damage (i.e. pollution, off-site impacts of soil erosion) to loss of either consumer welfare (i.e. through health impacts) or property and productivity (i.e. through damage to buildings, impacts on production).

$X_i \dots X_k$. Including mangrove area as a determinant of fish catch may therefore ‘capture’ some element of the economic contribution of this important ecological support function.

The above production function approach could be applied potentially to any of the various indirect use values of wetland systems indicated in Fig. 1. Thus this approach should prove to be a useful method of estimating these nonmarketed — but often significant — economic values. However, in order for this method to be applied, it is extremely important that the relationship between any environmental regulatory function and the economic activity it protects or supports is well understood.

Mäler (1991) distinguishes between applications of the production function approach. When production, Q , is measurable and either there is a market price for this output or one can be imputed, then determining the marginal value of the resource is relatively straightforward. If Q cannot be measured directly, then either a marketed substitute has to be found, or possible complementarity or substitutability between S and one or more of the other (marketed) inputs, $X_i \dots X_k$, has to be specified explicitly. Although all these applications require detailed knowledge of the physical effects on production of changes in the resource, S , and its environmental functions, applications that assume complementarity or substitutability between the resource and other inputs are particularly stringent on the information required on physical relationships in production. Clearly, cooperation is required between economists, ecologists and other researchers to determine the precise nature of these relationships.

Applications of the production function approach may be most straightforward in the case of single use systems, i.e. resource systems in which the predominant economic value is a single regulatory function, or a group of ecological functions providing support or protection for an economic activity in concert. In the case of multiple use systems — i.e. resource systems in which a regulatory function may support or protect many different economic activities, or which may have more than one regulatory ecological function of important economic value — applications of the production function approach may be slightly more problem-

atic. In particular, assumptions concerning the ecological relationships among these various multiple uses must be carefully constructed. Two major problems are double counting and trade offs between various direct and indirect use values, which appear whenever analysts attempt to aggregate the various direct and indirect use values arising from multiple use resource systems.

Aylward and Barbier (1992) provide an example of both on-site and off-site double-counting in terms of the nutrient retention function of a coastal wetland. Coastal wetlands often absorb organic nutrients from sewage and other waste emitted into waterways further upstream. Suppose that the nutrients held by the wetland are indirectly supporting both shrimp production within the wetland area and the growth of fish fry that supply an off-shore fishery. If the full value of the shrimp production is already accounted for as a direct use value of the wetland’s resources, adding in the share of the nutrient retention service as an indirect value and aggregating these values would double count this indirect use. In other words, the value of shrimp production already ‘captures’ the value-added contribution of nutrient retention.³ If instead one wanted to explicitly account for the value-added contribution to shrimp production of the nutrient retention function, then the direct value of the shrimp must be decreased to account for the return in value now attached to the nutrient retention service.

Similarly, if the fish fry supported through nutrients retained in the wetland eventually migrate to an off-shore fishery, then the indirect contribution during the fry’s stay in the wetland is included as an off-site component of the service’s value. That is, the nutrient retention function of the wetland produces an ‘external’ benefit in terms of supporting an off-shore fishery. Again, care must be taken to adjust the value of harvested fish in any companion analysis of the adjoining fishery to avoid misrepresenting the total economic value of the wetland and the fishery taken together.

Tradeoffs between two or more indirect use values of a given ecosystem may also occur. For example,

³ On the other hand, if the nutrient retention function of the wetland is valued only by its contribution to shrimp production, this function would be undervalued.

Barbier et al. (1993) illustrate why it is necessary to account for such trade-offs in their analysis of the Hadejia–Jama’are floodplain in northern Nigeria. The floodplain supports a number of important agricultural, forestry and fishing activities within the area of natural flooding. The floodplain also contributes to the recharge of groundwater, which is in turn drawn off by numerous small village wells throughout the region for domestic use and agricultural activities. However, concerns have recently been expressed about the excessive water use of pump-irrigated wheat production within the floodplain. Increasing use of the floodplain water to support this activity may mean less water available for natural groundwater recharge, and thus for village wells outside the floodplain. If there are tradeoffs between the two environmental support functions, then adding the full value of the wetland’s contribution to pump-irrigated wheat production within the floodplain to the full value of groundwater recharge of wells in neighbouring regions would overestimate the total benefit of these two environmental functions. In fact, in their analysis the authors had to adjust their estimates of the floodplain benefits for the ‘unsustainability’ of much pump-irrigated wheat production within the flooding area. The results of the analysis suggest that, even without considering the economic benefits of the groundwater recharge function, diverting water for upstream development does not make much economic sense if it is detrimental to the natural flooding system downstream.

Despite these pitfalls, many recent studies have attempted to employ the production function approach in valuing one or more regulatory functions of wetlands, in particular the role of estuarine wetlands and mangroves in supporting off-shore fisheries. There are two ways in which this approach has been implemented. The first is essentially a static approach, which either ignores the intertemporal fish harvesting process (i.e. assumes single-period or static production) or assumes that fish stocks are always constant (i.e. harvesting always offsets any natural growth in the fish population). Either assumption can be used to derive a market equilibrium for fish harvest, and thus to estimate changes in consumer and producer surplus arising from the impacts of a change in mangrove area on this static

equilibrium. The second is essentially a dynamic approach, which attempts to model the effects of a change in mangrove area on the growth function of the intertemporal fishing problem. By solving for the long-run equilibrium of the fishery, the comparative static effects and resulting welfare impacts of a change in mangrove area on the equilibrium levels of stock, effort and harvest can be determined.

3. Static models

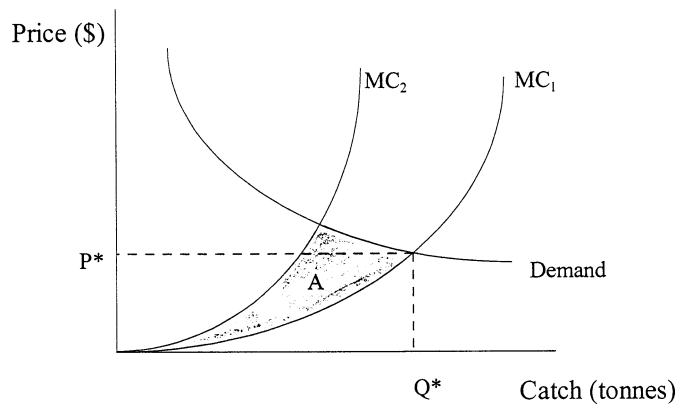
The static approach to valuing wetland-fishery linkages owes its development to a number of studies that have tried to determine the value of marshlands as habitat for Gulf Coast fisheries in the southern United States (Lynne et al., 1981; Ellis and Fisher, 1987; Farber and Costanza, 1987; Bell, 1989; Freeman, 1991; Bell, 1997).

The initial method was first developed by Lynne et al. (1981). Their approach was essentially half-way between the ‘static’ and ‘dynamic’ approaches described in this paper. Lynne et al. suggested that the support provided by the marshlands of southern Florida for the Gulf Coast fisheries could be modelled by assuming that marshland area is an additional argument in the bioeconomic growth equation of the fishery. Assuming that the latter function is logistic, and that harvesting of fish offsets any natural growth in fish stock, then the authors obtain the following relationship between fish harvest, h , fishing effort, E , and marshland area, M

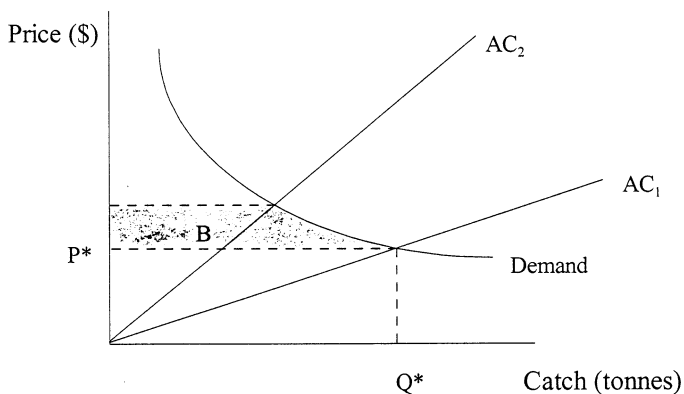
$$h_t = \beta_0 + \beta_1 E_t \ln M_{t-1} + \beta_2 E_t^2 \ln M_{t-1} + \mu_t \quad (2)$$

The parameters of Eq. (2) can be estimated from data on harvest, fishing effort and marshland area for those wetland-dependent species for which such data are available.⁴ Lynne et al. use such estimates and the price of harvested fish to derive the value

⁴ Note that Eq. (2) does not represent the complete long-run equilibrium of a typical intertemporal fishing model as the equation represents only one equilibrium condition, the bioeconomic condition of a constant level of fish stock. That is, because Eq. (2) excludes any consideration of price and costs in the determination of h , it does not represent the full long-run economic harvesting equilibrium of the fishery. For comparison, see the dynamic production function analysis discussed in the next sub-section.



The welfare impact of a change in wetland area on an optimally managed fishery is the change in consumer and producer surplus (area A) due to a shift in the marginal cost (MC) curve.



The welfare impact of a change in wetland area on an open access fishery is the change in consumer surplus (area B) due to a shift in the average cost (AC) curve.

Fig. 2. Welfare measures in optimally managed and open access fisheries in static models.

of the marginal productivity of a change in wetland area in terms of h . For example, for the blue crab fishery in western Florida salt marshes, the authors obtain a marginal productivity of 2.3 lb per year for each acre of marshland. Others have applied the Lynne et al. approach and Eq. (2) to additional Gulf Coast fisheries in western Florida (Bell, 1989, 1997; Farber and Costanza, 1987).

However, it is possible to view Eq. (2) as a kind of wetland-effort production function for a fishery, and assuming a static or one-period model,

one can employ this production function in a standard static optimization model of profit-maximizing harvesting behaviour. This is essentially the approach adopted by Ellis and Fisher (1987), who use the Lynne et al. (1981) case study to value the impacts of changes in the Florida Gulf Coast marshlands on the commercial blue crab fishery. Taking the sum of consumer and producer surplus as the measure of economic value, they hypothesize that an increase in wetland area increases the abundance of crabs and thus lowers

the cost of catch (see Fig. 2). The value of the wetlands' support for the fishery, which in this case is equivalent to the value of increments to wetland area, can then be imputed from the resulting changes in consumer and producer surplus.

An important assumption in the Ellis and Fisher model is that Lynne et al.'s Eq. (2) can be approximated by the Cobb–Douglas form

$$h = AE^\alpha M^\beta \quad (3)$$

where h is the quantity of crab catch in pounds, E is catch effort measured by traps set and M is area of wetlands. From the profit-maximizing conditions of the static optimization model for harvesting, the corresponding cost function, C , is

$$C = WA^{-1/\alpha} M^{-\beta/\alpha} h^{1/\alpha} \quad (4)$$

where W is the unit cost of effort. Assuming an iso-elastic demand for crabs and either private ownership or optimal public management (i.e. price equals marginal cost in both cases), Ellis and Fisher are able to estimate the change in consumer and producer surplus in the market for blue crabs resulting from a change in marshland area (see Fig. 2).

Freeman (1991) extends further Ellis and Fisher's approach to show how the values imputed to the wetlands are influenced by the market conditions and regulatory policies that affect harvesting decisions in the fishery, in particular whether it operates under conditions of open access or optimal management. For example, under open access, rents in the fishery would be dissipated, and price would be equated to average and not marginal costs. As a consequence, producer surplus is zero and only consumer surplus determines the value of increased wetland area (see Fig. 2). Freeman demonstrates that when the demand for crabs is inelastic, the social value of an increase in area is higher under open access than under optimal regulation, whereas the wetlands are more valuable under optimal regulation when demand is elastic. This result stems from the role of price changes in allocating welfare gains between producers and consumers: in the case of optimal regulations, part of the consumers' gain is a transfer from producer surplus, whereas under open

access and zero producer surplus, any reduction in the price of fish associated with the average cost curve shifting down (in response to an increase in wetland area) results in a gain in consumer surplus and increased wetland value.

Freeman also calculates the social value of the marginal product of marshland area, VMP_M , which from Eq. (3) is

$$VMP_M = P\beta \frac{h}{M} \quad (5)$$

where P is the price of crabs. As optimal regulation should lead to a higher price than open access, an inelastic demand means that VMP_M is higher under optimal regulation.

These different impacts of market conditions and regulatory policies for the production function approach to valuing biological resources and systems, where open access exploitation and imperfect markets for resources are common. As argued by Barbier (1994), this may be a prevalent feature of resource systems in tropical regions.

Applications of the production function approach to value more than one regulatory function of a wetland that supports or protects many different economic activities are rare. As noted above, assumptions concerning the ecological relationships among these various multiple uses must be carefully constructed, and the data for this analysis are often not available.

For example, Ruitenbeek (1994) uses a modified production function approach to evaluate the trade-offs between different forestry options for a mangrove system in Bintuni Bay, Irian Jaya, Indonesia. The options range from preserving the mangroves through a cutting ban to various forestry development options involving partial, selective and clear cutting operations. An important feature of the analysis is that tries to incorporate explicitly the linkages between loss of mangroves and their ecological functions and the productivity of economic activities. For example, the mangroves may support many economic activities, such as commercial shrimp fishing, commercial sago production and traditional household production from hunting, fishing, gathering and cottage industry; they may also have an indirect use value through controlling erosion and sedi-

mentation, which protects agricultural production in the region; and they have an indirect role in supporting biodiversity. To the extent that the ecological linkages in terms of support or protection of these activities are strong, then the opportunity cost of forestry options that lead to the depletion or degradation of the mangroves will be high. Thus, the ‘optimal’ forest management option — whether clear cutting, selective cutting or complete preservation — depends critically on the strength of the ecological linkages.

In the absence of any ecological data on these linkages, Ruitenbeek develops several different scenarios based on different linkage assumptions. This essentially amounted to specifying more specifically the relationship between Q and S in the simple production function relationship Eq. (1) indicated above. Thus for each productive activity at time t , Q_{it} , the following relationship is assumed

$$Q_{it}/Q_{i0} = (S_t - \tau/S_0)^\alpha \quad (6)$$

where S_t is the area of remaining undisturbed mangroves at time t , α and τ are impact intensity and delay parameters, respectively, $Q_{i0} = Q_{it}(t = 0)$ and $S_0 = S_t(t = 0)$. For example, for fishery-mangrove linkages, a moderate linkage of $\alpha = 0.5$ and $\tau = 5$ would imply that shrimp output varies with the square root of mangrove area (e.g. a 50% reduction in mangrove area would result in a 30% fall in shrimp production), and there would be a delay of 5 years before the impact takes effect. If no ecological linkages are present, i.e. there is no indirect use value of mangroves in terms of supporting shrimp fishing, then $\alpha = 0$. At the other extreme, very strong linkages imply that the impacts of mangrove removal are linear and immediate, i.e. $\alpha = 1$ and $\tau = 0$.

Based on his analysis, Ruitenbeek concludes that the assumption of no or weak environmental linkages is unrealistic for most economic activities related to the mangroves. Moreover, given the uncertainty over these ecological linkages and the high costs associated with irreversible loss, if environmental linkages prove to be significant, then only modest selective cutting (e.g. 25% or less) of the mangrove area was recommended.

4. Dynamic models

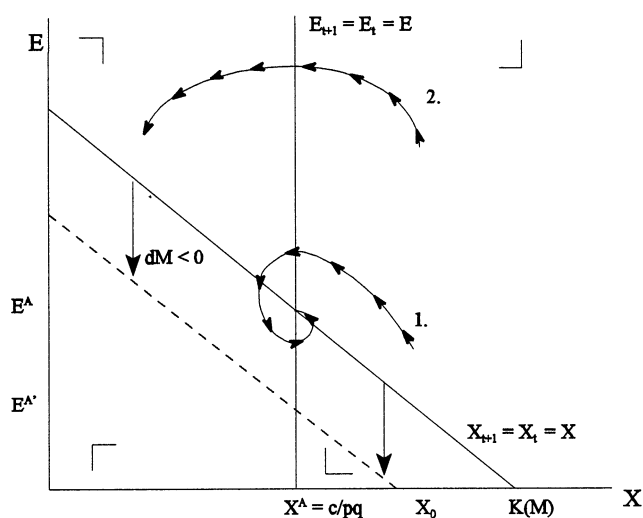
The production function approach can also be incorporated into intertemporal models of renewable resource harvesting in cases where the ecological function affects the growth rate of a stock over time. In such cases, the production function link is a dynamic one, as the ecological function affects the rate at which a renewable resource increases over time, which in turn affects the amount of offtake, or harvest, of the resource. The basic approach to valuation of an environmental input to renewable resource production in a dynamic context is outlined by Hammack and Brown (1974), Ellis and Fisher (1987), Freeman (1993), Barbier and Strand (1998).

As shown by Barbier and Strand (1998), adapting bioeconomic fishery models to account for the role of a mangrove system in terms of supporting the fishery as a breeding ground and nursery habitat is fairly straightforward, if it is assumed in the fishery model that the effect of changes in mangrove area is on the carrying capacity of the stock and thus indirectly on production.⁵ Defining X_t as the stock of fish measured in biomass units, any net change in growth of this stock over time can be represented as

$$X_{t+1} - X_t = F(X_t, M_t) - h(X_t, E_t), \quad F_X > 0, \quad F_M > 0 \quad (7)$$

Thus net expansion in the fish stock occurs as a result of biological growth in the current period, $F(X_t, M_t)$, net of any harvesting, $h(X_t, E_t)$. Note that the standard fish harvesting function is employed; i.e. harvesting is a function of the stock as well as fishing effort, E_t . Instead, it is the biological growth function of the fishery that is modified to allow for the influence of mangrove area, M_t , as a breeding ground and nursery. It is reasonable to assume that this influence on growth is positive, i.e. $\partial F/\partial M_t = F_M > 0$, as an increase in mangrove area will mean more carrying capacity for the fishery and thus greater biological growth.

⁵ For analytical convenience, a discrete time model of the fishery is employed here.



The effect of a fall in mangrove area, M , on the long-run equilibrium of an open access fishery is to reduce the steady state level of effort, E^A . Since harvesting is an increasing function of effort, long-run harvesting output in the fishery will also fall.

Source: Barbier and Strand (1998).

Fig. 3. Mangrove loss and the long-run equilibrium of an open access fishery.

Eq. (7) can now be employed in a standard intertemporal harvesting model of the fishery, where depending on the management regime, harvesting over time can either be depicted to occur under open access conditions (i.e. effort in the fishery adjusts over time to the availability of profits) or under optimal management conditions (the discounted net returns from harvesting the fishery are maximized over time). The effect of a change in mangrove area can therefore be valued in terms of changes in the optimal path of harvesting over the period of analysis and in terms of the changes in the long-run equilibrium of the fishery.

Fig. 3 shows the fairly straightforward case analyzed by Barbier and Strand, where the effects of a change in mangrove area is depicted in terms of influencing the long-run equilibrium of an open access fishery. In the figure, the long-run equilibrium of the fishery is depicted in terms of steady values for effort, E , and fish stocks, X . As discussed above, the carrying capacity of the fishery is assumed to be an increasing function of mangrove area, i.e. $K = K(M)$, $K_M > 0$. Trajectory one

shows an optimal path to a stable long-run equilibrium for the fishery. In this case, a decrease in mangrove area causes the long-run level of fishing effort to fall. As harvesting levels are generally positively related to effort levels, the consequence of mangrove deforestation is also a decrease in equilibrium fish harvest.

5. A case study of a static model: southern Thailand

Sathirathai (1997) uses the Ellis-Fisher-Freeman model to value the welfare impacts of mangrove deforestation on coastal fisheries in Surat Thani Province on the Gulf of Thailand. In recent decades, the expansion of intensive shrimp farming in the coastal areas of southern Thailand has led to rapid conversion of mangroves. Over 1975–1993 the area of mangroves has virtually halved, from 312 700 hectares (ha) to 168 683 ha. Although the rate of mangrove loss has slowed, in the early 1990s the annual loss was estimated to

Table 1

Welfare estimates of changes in mangrove area on the Gulf of Thailand shellfish and demersal fisheries^a

Management regime	Economic value of a change in mangrove area (US\$ per ha) ^b			
	Demand elasticity	Demersal fish	Shellfish	All fish
<i>(Open access)</i>				
	$\eta = -0.1$	63.48	46.75	110.23
	$\eta = -1$	39.71	43.29	83.00
	$\eta = -10$	8.38	24.92	33.30
<i>(Managed fisheries)</i>				
	$\eta = -0.1$	38.74	44.47	83.21
	$\eta = -1$	38.88	44.50	83.38
	$\eta = -10$	39.06	44.63	83.69
Economic value of annual loss of 1200 ha of mangrove area (US\$) ^c				
<i>(Open access)</i>				
	$\eta = -0.1$	76 176	56 100	132 276
	$\eta = -1$	47 652	51 948	99 600
	$\eta = -10$	10 056	29 904	39 960
<i>(Managed fisheries)</i>				
	$\eta = -0.1$	46 488	53 364	99 852
	$\eta = -1$	46 656	53 400	100 056
	$\eta = -10$	46 872	53 556	100 428

^a Based on Sathirathai (1997).^b Calculations assume an initial equilibrium quantity demand and price based on observed data for Surat Thani Province (Zone three) in 1993. For demersal fish this is harvested output of 1 545 000 kg and price of US\$ 1.51/kg, and for shellfish 1 917 000 kg and US\$2.58/kg.^c Over 1991–3, the average annual loss of mangroves in Surat Thani province (Zone three) was estimated to be around 12.19 km², or around 1200 hectares (ha).

be around 3000 ha/year for all of Thailand, and 1200 ha/year in Surat Thani province.

The Gulf of Thailand mangroves are thought to provide breeding grounds and nurseries in support of several species of demersal fish and shellfish, mainly crab and shrimp.⁶ To analyze the impact of mangrove deforestation on these fisheries in Surat Thani, Sathirathai assumes that harvesting in both demersal and shellfish fisheries is a Cobb–Douglas function of the level of fishing effort and mangrove area, as depicted by Eq. (3).⁷

⁶ Mangrove-dependent demersal fish include those belonging to the *Clupeidae*, *Chanidae*, *Ariidae*, *Plotosidae*, *Mugilidae*, *Lujanidae* and *Latidae* families. The shellfish include those belonging to the families of *Panaeidae* for shrimp and *Grapsidae*, *Ocypodidae* and *Portunidae* for crab.

⁷ In this study, total fishing effort per year is the number of fishing instruments (e.g. gill net boats) recorded per annum times the average of hours spent on fishing per fishing instrument each year.

A separate harvesting function is assumed to apply to demersal fish as opposed to shellfish.

Sathirathai conducts a panel analysis to estimate a log-linear version of Eq. (3) for all shellfish and all demersal fish in the Gulf of Thailand. The analysis combines harvesting, effort and mangrove data across all five zones of the Gulf of Thailand and over the 1983–93 time period. This allows estimation of the parameters A , α and β in Eq. (3), for two separate Cobb–Douglas production functions, one each for demersal fish and shellfish. Combining this information with the estimated unit cost of effort, W , allows the Cobb–Douglas cost function Eq. (4) to be specified for both demersal fish and shellfish for each of these fisheries in Surat Thani Province. This province is an important fishing region in Zone three of the Gulf of Thailand. Following the methodology indicated in Fig. 2, Sathirathai uses the cost functions derived for each fishery to

estimate the likely welfare impacts of a change in mangrove area in Surat Thani, assuming alternatively open access and managed fishery conditions.

Table 1 shows the results of the welfare calculation for the impact of a per ha change in mangrove area on the Gulf of Thailand shellfish and demersal fisheries of Surat Thani Province. For all mangrove-dependent fisheries, the value of a change in mangrove area ranges from US\$33–110/ha, depending on whether the fisheries are open access or managed. Similar to the outcome reported by Freeman (1991) for the Florida Gulf Coast blue crab fishery, when the demand for Gulf of Thailand fish is inelastic, the value of a change in mangrove area in Surat Thani is higher under open access than under optimal regulation, whereas the wetlands are more valuable under optimal regulation when demand is elastic. Under managed fishery conditions, different demand elasticity assumptions hardly affect the welfare estimates of a change in mangrove area, which are estimated to be around US\$39/ha for demersal fish and US\$45/ha for shellfish. In the open access scenario, changes in elasticities affect more the value of mangroves in supporting demersal fisheries as opposed to shellfish. Thus under open access and depending on the elasticity of demand, the value of the mangroves in Surat Thani ranges from US\$8–63/ha for demersal fish and from US\$25–47/ha for shellfish.

Table 1 also shows the estimated welfare impacts associated with the annual loss of 1200 ha of mangroves in Surat Thani, which was approximately the annual rate of mangrove conversion recorded in the early 1990s in the province. Given this rate of deforestation, the economic loss in terms of support of the Gulf of Thailand fisheries in Zone three is estimated to be around US\$100 000 per year, if these fisheries were optimally managed. Under open access conditions, this economic loss ranges from US\$40 000 to 132 000, depending on demand elasticities.

6. A case study of a dynamic model: Campeche, Mexico

Barbier and Strand (1998) employ the dynamic approach to production function analysis to value

the role of mangroves in the Laguna de Terminos in supporting the shrimp fishery of Campeche, Mexico.

Mexico's gulf coast states account for over half of the country's shrimp catch, and the state of Campeche is responsible for one-sixth of Mexico's total output of shrimp. Campeche's shrimp fishery employs about 13% of the state's economically active population. In recent years the total number of boats in the fishery have increased substantially, but the composition of the fleet has also changed significantly. There has been a substantial decline in the number of commercial vessels, whereas the artisanal fleet has expanded rapidly. From 1980–1987, production in the shrimp fishery fluctuated steadily between 7–8 thousand metric tons (KMT), but by 1990 output had fallen to 4.6 KMT.

The mangroves in the Laguna de Terminos are considered by ecologists to be the main breeding ground and nursery habitat for the shrimp fry of the Campeche fishery (Yañez-Arancibia and Day, 1988). Mangrove area was estimated to be around 860 km² in 1980, declining to about 835 km² in 1991, a loss of around 2 km² per annum. The primary reason for the loss is the encroachment of population from Carmen, the large city adjacent to Laguna de Terminos. Future threats are expected to come from expansion of shrimp aquaculture through conversion of coastal mangroves, and possibly pollution.

Barbier and Strand model the effects of mangrove deforestation in Laguna de Terminos by use of comparative static analysis of the long-run equilibrium, as depicted in Fig. 3. In their model of the Campeche shrimp fishery, they assume that the basic growth function of the fishery is logistic and that shrimp harvesting follows the conventional Schaefer production process, $h_t = qE_tX_t$. Thus Eq. (7) becomes

$$X_{t+1} - X_t = [r(K(M_t) - X_t) - qE_t]X_t \quad (8)$$

where r is the intrinsic growth of shrimp each period, K is the environmental carrying capacity of the system and mangrove area, M_t , has a positive impact on carrying capacity, i.e. $K_M > 0$.

To estimate the comparative static effects of a change in mangrove area on long-run shrimp harvesting, Barbier and Strand assume a propor-

tional relationship between mangrove area and carrying capacity, i.e. $K(M) = \alpha M$, $\alpha > 0$. As the shrimp stock is constant in the long-run equilibrium, $X_t = X_{t+1} = X$, then using this condition in Eq. (8) and the Schaefer production function to substitute for X , the following relationship between shrimp production, mangrove area and effort is derived

$$h = qEK(M) - \frac{q^2}{r}E^2 = q\alpha EM - \frac{q^2}{r}E^2 \quad (9)$$

The authors estimate Eq. (9) by employing 1980–90 time series data on shrimp harvests, effort and mangrove area for Campeche, Mexico, to derive the parameters $b_1 = \alpha q$ and $b_2 = -q^2/r$.

A second condition of the long-run equilibrium of an open access fishery is that profits will be zero, i.e. $ph = cE$, where p is the price of shrimp catch and c is the cost of fishing effort. In order to simulate the comparative static effects of a change in mangrove area, Barbier and Strand assume that this ‘zero profit’ condition holds for the Campeche shrimp fishery. Using actual price data on shrimp catches over this period, they calculate the costs of effort, c^A , necessary for the zero profit condition to hold for the Campeche fishery over 1980–90. Using the estimated parameters of Eq.

(9) with the price and cost data, the authors simulate the effects of a change of mangrove area on equilibrium harvesting and gross revenues in the Campeche shrimp fishery over 1980–90.

The results are depicted in Table 2. On average over the 1980–90 period, a marginal (in km²) decline in mangrove area produces a loss of about 14.4 metric tons of shrimp harvest and nearly US\$140 000 in revenues from the Campeche fishery each year. However, given the relatively small rate of annual mangrove deforestation in the region over the 1980–90 period — around 2 km² per year — the resulting loss in shrimp harvest and revenues does not appear to be substantial, only around 0.4% per year.

The simulation in Table 2 also demonstrates how the economic losses associated with mangrove deforestation are affected by long-run management of the open access fishery. As noted above, the early years of the period of analysis (e.g. 1980–81) were characterized by much lower levels of fishing effort and higher harvests (e.g. on average around 4800 combined vessels extracting about 8.5 KMT annually). Table 2 shows that, if this earlier period represented the open access equilibrium of the fishery, the economic impacts of a marginal (km²) decline in mangrove area

Table 2
Simulation estimates of a marginal change in mangrove area, Campeche, Mexico (d M)^{a,b}

Year	Price (p) US\$/kg ^c	Cost (c^A) US\$/vessel ^d	Change in equilibrium harvest (d h^A) metric tons	Change in equilibrium revenues (p d h^A) US\$	Change %
1980	7.10	13 984	20.40	144 808	0.23
1981	9.68	15 628	16.72	161 826	0.20
1982	10.57	13 816	13.53	143 060	0.18
1983	9.80	13 636	14.41	141 197	0.18
1984	9.83	14 096	14.85	145 963	0.19
1985	9.80	16 687	17.63	172 798	0.20
1986	10.00	15 013	15.55	155 460	0.19
1987	10.22	14 363	14.55	148 731	0.20
1988	10.56	14 132	13.86	146 334	0.20
1989	10.21	10 000	10.14	103 547	0.17
1990	10.40	6677	6.65	69 143	0.14
Mean	9.83	13 457	14.39	139 352	0.19

^a Source: Barbier and Strand (1998).

^b Parameter estimates: $b_1 = 4.4491$; $b_2 = -0.4297$.

^c US\$/kg, in real (1982) prices.

^d c^A is the ‘equilibrium’ (real) cost per unit effort, defined as the cost level necessary to attain zero profit in the fishery, i.e. $c^A = ph^A/E^A$.

would be a reduction in annual shrimp harvests of around 18.6 tons, or a loss of about US\$153 300 per year. In contrast, the last two years of the analysis (e.g. 1989–90) saw much higher levels of effort and lower harvests in the fishery (e.g. around 6700 combined vessels extracting 5.3 KMT annually). As a consequence, if this latter period represents the open access equilibrium, then a marginal decline in mangrove area would result in annual losses in shrimp harvests of 8.4 tons, or US\$86 345 each year.

Thus, the value of the Laguna de Terminos mangrove habitat in supporting the Campeche shrimp fishery appears to be affected by the level of exploitation. This suggests that, if an open access fishery is more heavily exploited in the long run, the subsequent welfare losses associated with the destruction of natural habitat supporting this fishery are likely to be lower. Intuitively, this makes sense. The economic value of an over-exploited fishery will be lower than if it were less heavily depleted in the long run. The share of this value that is attributable to the ecological support function of natural habitat will therefore also be smaller.

The management implications are clear: As long as effort levels continue to rise, harvests will fall, even if mangrove areas are fully protected. Moreover, any increase in harvest and revenues from an expansion in mangrove area is likely to be short-lived, as it would simply draw more effort into the fishery. Better management of the Campeche shrimp fishery to control over-exploitation may be the only short-term policy to bring production back to respectable levels, as well as realizing the more long-term economic benefits of protecting mangrove habitat.

7. Conclusion

This paper has indicated how the economic value of mangroves in supporting coastal and marine fisheries can be estimated through application of production function approaches. Both basic static and dynamic production function models for estimating this value have been reviewed. Case studies of the application of the static and dy-

namic approaches to valuing the support function of mangroves in Thailand and Mexico have also been examined.

The production function approach appears to be well suited to valuing the important ecological role of coastal and estuarine wetlands in supporting offshore fisheries. As these wetland systems are under considerable threat from coastal development, it is important to develop reliable economic estimates of the value of their ecological support function role. Failure to consider this value may misrepresent the economic costs associated with wetland conversion, which are too often assumed to be insignificant or zero in coastal development decisions. This is particularly the case in developing countries, where many mangrove systems are threatened with conversion through the expansion in coastal areas of aquaculture, agriculture, tourism, and urban and infrastructural development.

However, both static and dynamic models show that, in applying the production function approach to valuing the support of wetlands for offshore fisheries, any resulting welfare estimate will be affected significantly by whether the fisheries are managed or subject to open access. For example, the Gulf of Thailand study indicates that static production function estimates of the value of a change in mangrove area in terms of support of managed fisheries will be little affected by different demand elasticities. In contrast, for open access fisheries, the value of the mangrove support function will tend to be much lower for elastic as opposed to inelastic market demand for harvested fish. In the case of dynamic models of mangrove-fishery linkages, the Mexican case study illustrates how the economic losses associated with mangrove deforestation are influenced by the long-run management conditions in the open access fishery. That is, if an open access fishery is more heavily exploited in the long run, the subsequent welfare losses associated with any mangrove habitat supporting this fishery are likely to be lower, as the mangroves will now be supporting a more over-exploited and thus less valuable fishery.

The methodologies and case studies discussed in this paper show the important potential in utilizing production function approaches to valuing the

environment as input, particularly valuing the ecological support functions of wetlands, such as mangrove systems. Ecologists have indicated that the regulatory functions performed by wetlands and other complex natural ecosystems may be highly significant in supporting and protecting economic activity. Perhaps the next phase in the development of production function approaches will be to apply such methodologies not just to valuing single-use functions of wetlands, such as the role of mangroves as nursery and breeding ground habitats for coastal and marine fisheries, but also to valuing simultaneously the diverse range of regulatory functions typically found in a multi-use natural wetland, such as those listed in Fig. 1.

Acknowledgements

A version of this paper was prepared for the 4th Workshop of the Global Wetlands Economics Network (GWEN), Wetlands: Landscape and Institutional Perspectives, Beijer International Institute of Ecological Economics, The Royal Swedish Academy of Sciences, Stockholm, Sweden, 16–17 November 1997. I am grateful to Jack Ruitenbeek, Tore Söderquist and an anonymous referee for helpful comments. However, the usual caveat applies.

References

- Aylward, B.A., Barbier, E.B., 1992. Valuing environmental functions in developing countries. *Biodivers. Conserv.* 1, 34–50.
- Barbier, E.B., 1994. Valuing environmental functions: tropical wetlands. *Land Econom.* 70, 155–173.
- Barbier, E.B., Acreman, M., Knowler, D., 1997. *Economic Valuation of Wetlands: A Guide for Policymakers*. Ramsar Convention Bureau, Geneva, 127 pp.
- Barbier, E.B., Adams, W.M., Kimmage, K., 1993. An economic valuation of wetland benefits. In: Hollis, G.E., Adams, W.M., Aminu-Kano, M. (Eds.), *The Hadejia–Nguru Wetlands: Environment, Economy and Sustainable Development of a Sahelian Floodplain Wetland*. IUCN, Geneva, pp. 191–209.
- Barbier, E.B., Strand, I., 1998. Valuing mangrove-fishery linkages: a case study of Campeche, Mexico. *Environ. Resour. Econ.* 12, 151–166.
- Bockstael, N.E., McConnell, K.E., 1981. Theory and estimation of the household production function for wildlife recreation. *J. Environ. Econ. Manag.* 8, 199–214.
- Bell, F.W., 1989. *Application of Wetland Valuation Theory to Florida Fisheries*. Report No. 95, Florida Sea Grant Program, Florida State University, Tallahassee.
- Bell, F.W., 1997. The economic value of saltwater marsh supporting marine recreational fishing in the Southeastern United States. *Ecol. Econ.* 21, 243–254.
- Ellis, G.M., Fisher, A.C., 1987. Valuing the environment as input. *J. Environ. Manag.* 25, 149–156.
- Farber, S., Costanza, R., 1987. The economic value of wetlands systems. *J. Environ. Manag.* 24, 41–51.
- Freeman, A.M., 1991. Valuing environmental resources under alternative management regimes. *Ecol. Econ.* 3, 247–256.
- Freeman, A.M., 1993. *The Measurement of Environmental and Resource Values: Theory and Methods*. Resources for the Future, Washington DC, pp. 516.
- Hammack, J. Brown, G.M. Jr., 1974. *Waterfowl and Wetlands: Towards Bioeconomic Analysis*. Resources for the Future, Washington DC, pp. 273.
- Kahn, J.R., Kemp, W.M., 1985. Economic losses associated with the degradation of an ecosystem: the case of submerged aquatic vegetation in Chesapeake Bay. *J. Environ. Econ. Manag.* 12, 246–263.
- Knowler, D., Strand, I., Barbier, E.B., 1997. *An Economic Analysis of Black Sea Fisheries and Environmental Management*. Final Report, The Black Sea Environment Programme, The World Bank/UNEP Global Environmental Facility, Istanbul, Rome.
- Lynne, G.D., Conroy, P., Prochaska, F.J., 1981. Economic value of marsh areas for marine production processes. *J. Environ. Econ. Manag.* 8, 175–186.
- Mäler, K.-G., 1991. The production function approach. In: Vincent, J.R., Crawford, E.W. and Hoehn, J.P. (Eds.) *Valuing Environmental Benefits in Developing Countries*. Special Report 29. Michigan State University, East Lansing, pp. 11–32.
- Parks, P., Bonifaz, M., 1994. Nonsustainable use of renewable resources: mangrove deforestation and mariculture in Ecuador. *Mar. Resour. Econ.* 9, 1–18.
- Ruitenbeek, H.J., 1994. Modelling economy-ecology linkages in mangroves: economic evidence for promoting conservation in Bintuni Bay, Indonesia. *Ecol. Econ.* 10, 233–247.
- Sathirathai, S., 1997. *Economic Valuation of Mangroves and the Roles of Local Communities in the Conservation of the Resources: Case Study of Surat Thani, South of Thailand*. Final report submitted to the Economy and Environment Program for Southeast Asia (EEPSEA), EEPSEA, Singapore.
- Smith, V.K., 1991. Household production functions and environmental benefit estimation. In: Braden, J.B., Kolstad, C.D. (Eds.), *Measuring the Demand for Environmental Quality*. North-Holland, Amsterdam, pp. 41–76.
- Strand, I.E., Bockstael, N.E., 1990. Interaction between agriculture and fisheries: empirical evidence and policy implications. In: Just, R.E., Bockstael, N.E. (Eds.), *Commodity*

- and Resource Policies in Agricultural Systems. Springer-Verlage, New York, pp. 50–61.
- Swallow, S.K., 1990. Depletion of the environmental basis for renewable resources: the economics of interdependent renewable and nonrenewable resources. *J. Environ. Econ. Manag.* 19, 281–296.
- Swallow, S.K., 1994. Renewable and nonrenewable resource theory applied to coastal agriculture, forest, wetland and fishery linkages. *Mar. Resour. Econ.* 9, 291–310.
- Yañez-Arancibia, A., Day, J.W. Jr. (Eds.), 1988. *Ecology of coastal ecosystems in the southern gulf of Mexico: The Terminos Lagoon Region*. UNAM Press, Mexico, 518 pp.