ORIGINAL ARTICLE

Supply-side approaches to the economic valuation of coastal and marine habitat in the Red Sea

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Abstract   The degradation of natural fish habitat in the ocean implies lost economic benefits. These value losses often are not measured or anticipated fully, and therefore they are mainly ignored in decisions to develop the coast for industrial or residential purposes. In such circumstances, the ocean habitat and its associated ecosystem are treated as if they are worthless. Measures of actual or potential economic values generated by fisheries in commercial markets can be used to assess a conservative (lower-bound) value of ocean habitat. With this information, one can begin to compare the values of coastal developments to the values of foregone ocean habitat in order to help understand whether development would be justified economically. In this paper, we focus on the economic value associated with the harvesting of commercial fish stocks as a relevant case for the Saudi Arabian portion of the Red Sea. We describe first the conceptual basis behind supply-side approaches to economic valuation. Next we review the literature on the use of these methods for valuing ocean habitat. We provide an example based on recent research assessing the bioeconomic status of the traditional fisheries of the Red Sea in the Kingdom of Saudi Arabia (KSA). We estimate the economic value of ecosystem services provided by the KSA Red Sea coral reefs, finding that annual per-unit values supporting the traditional fisheries only are on the order of $7000/km^2. Finally, we develop some recommendations for refining future applications of these methods to the Red Sea environment and for further research.

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1. Introduction

Humans depend upon the natural resources of the Red Sea, including its fish stocks (PERSGA, 2002). Red Sea fish stocks are exploited for both subsistence and commerce. They may also be used for recreation (Gladstone et al., 2012). Economic value is generated by all of these activities.

While humans have been living and using the resources of the Red Sea coast for many millennia, development has become more extensive–even industrialized–in recent years, especially along the Red Sea coast of the Kingdom of Saudi Arabia

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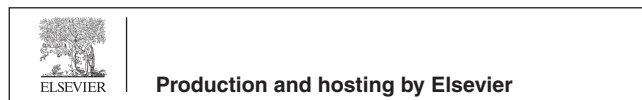
(KSA). In some cases, this development has led to the degradation of natural fish habitat, including the loss of mangrove wetlands, seagrass beds, and coral reefs (Kotb, 2010; Gladstone et al., 2006). Moreover, coastal development can lead to increased sedimentation due to erosion and to nutrient releases from industrial effluents or sewage disposals. Sediments and nutrient loads affect water quality adversely, thereby further degrading fish habitat (El Sayyed, 2008).

The degradation or loss of natural fish habitat implies losses in economic values. Commercial, subsistence, and recreational fisheries are all affected adversely. In most cases, this loss in value is not measured or anticipated fully, and therefore it is often ignored in decisions to develop the coast. In effect, the ocean habitat, here including coastal wetlands, the seabed and its flora, coral reefs, ocean waters, and the associated ecosystem, is treated as if it may be worthless (*i.e.*, as if it has no price).

Importantly, measures of the economic value generated by fisheries can be used to estimate the value of the ocean habitat that supports fish stocks. These measures are commonly known as "supply-side" or "productivity" approaches to habitat valuation (Barbier, 2007; McConnell and Bockstael, 2005). These measures can be utilized to impute a conservative (lower bound) value on ocean habitat. (This value is conservative because ocean habitat may be a source of value for other ocean uses, including recreation and passive, non-market benefits.) With this information, the value of coastal development can be compared to the value of potentially foregone ocean habitat in order to determine whether the development would be justified economically.

In this paper, we focus on the economic value associated with the exploitation of commercial fish stocks as the most relevant case for the Saudi Arabian portion of the Red Sea. Similar methods may apply to recreational fisheries (Bell, 1997) or, more broadly, to other uses of the ocean or functions of habitat (Barbier et al., 2011). The precise sources and measures of economic value may depend upon the type of use under consideration, however (Barbier et al., 2008).

We describe first the conceptual basis behind supply-side approaches to valuation. Next we review the literature on the use of these methods for valuing ocean habitat. We consider an example based on recent research assessing the bioeconomic status of the traditional fisheries of the Red Sea in the Kingdom of Saudi Arabia. Finally, we develop some recommendations for the application of these methods to the Saudi Arabian Red Sea environment.

2. Commercial fisheries and resource rents

The commercial harvesting of fish stocks results in the production of seafood as an economic commodity. Using a commercial fishing technology, sometimes referred to as a "black box," fishermen produce seafood by combining factors of production, including labor, experience and knowledge, fishing vessels, fuel, nets, bait, ice, and other inputs (Fig. 1). Fish as seafood may be sold as it is or processed for value-added.[1]

Economic surpluses result from the harvest of fish stocks. In a well-managed fishery, these surpluses are distributed be-

Figure 1   Economic and ecological production functions.
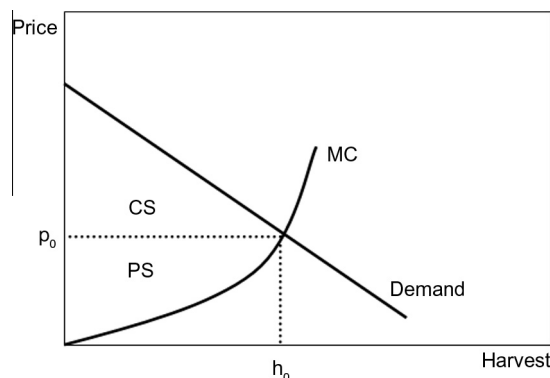
Figure 2a   *Well-managed* (rationalized) fishery with price equal to marginal cost (MC). The market equilibrium occurs at price, p_0, and harvest, h_0. Both consumer surplus (CS) and producer surplus (PS) are realized. If all fishing vessels are identical, in terms of both capital and labor (including fishing skills and knowledge), then the producer surplus comprises the resource rent.

tween fishermen (as producers) and seafood consumers. Taken together, producer and consumer surpluses are the economic measure of value for a commercial fishery (Fig. 2a). Producer surplus is equivalent to the revenues earned from selling catch net of all costs of fishing. Producer surplus is represented by the area below price and above the supply schedule. Consumer surplus is evaluated as what consumers are willing to pay for seafood, less what they actually pay in the market. Consumer surplus is represented by the area below the demand schedule and above price.

One element of the producer surplus is known as the "resource rent." Resource rent is the cost of fish utilized as an input in the production of seafood as a commodity. Resource rent implies that fish have a price, although nature does not charge fishermen this price when fish are removed from their habitat. Because fish both grow and reproduce, the removal of fish from the ocean by harvesting imposes a dynamic cost that depends upon the size of the relevant population; it is this "user cost" that comprises resource rent.[2] Because this price is

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[1] Artisanal fishermen may harvest fish for subsistence purposes. Although such fish do not enter a formal market, they are still to be regarded as an economic commodity.
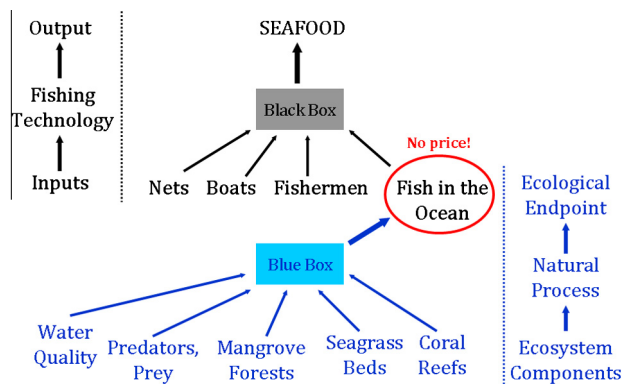
[2] The resource rent can be interpreted as the value of the ecosystem "service" as embodied in wild fish stocks, *per se* (see below). This review is focused more generally on measures of the value of the ecosystem or habitat that supports wild fish stocks.

not exacted by nature, and it must be estimated by humans, it is sometimes referred to as a "shadow price."

A fish stock, X, is a renewable resource. The fish stock is capable of growth, *i.e.*, there is a dynamic "flow" of fish, where $\dot{X}$ denotes the change in the stock per unit of time, which depends upon the size of the stock at any point in time:

$$\dot{X} = F(X)$$

Typically the growth of the fish stock, $F(X)$, increases at low stock sizes, reaches a peak at a level known as "maximum sustainable yield" (MSY), and decreases until the stock reaches its ecological "carrying capacity." Fish harvest, $h(E,X)$, is a function of both fishing effort, E,[3] and stock size, X, and it removes biomass from the stock:

$$\dot{X} = F(X) - h(E, X)$$

Fish may be harvested from the ecosystem at a cost, $c(X)$, that depends inversely upon the size of the stock. Fish that have been harvested are delivered to a market, where they fetch a price, $p(h)$, which depends upon the demand for and the supply of fish. In steady-state equilibrium, or at any point in time on a path to equilibrium, it is economically optimal to harvest the fish stock so that the discounted sum of net social benefits (*i.e.*, economic surpluses) per period is maximized. Under the optimal condition, the marginal revenue of harvest equals the marginal cost of harvest, $c(X)$, plus the user cost (the resource rent), μ. Thus, for a constant price of fish at the dock, p, the resource rent is:

$$\mu = p - c(X)$$

At the economic optimum, fish should be harvested according to the following "modified golden rule":

$$\delta = F_X - \frac{c_X F}{\mu}$$

This rule states that the difference between an incremental "investment" in the fish stock *in situ*, F_X (by letting it continue to grow in the ocean), and the incremental gain from harvesting it now, $c_X F / \mu$, must equal the discount rate, δ (Munro and Scott, 1985).

Because there is no market for fish prior to harvest in their natural environment, in an unregulated fishery, fishermen treat fish as an unpriced input in the production of seafood (Fig. 1). Fishermen fish more than they should because the absence of a price on fish in the ocean is perceived as an implicit subsidy in the production of seafood. In such a situation, competition among fishermen for the fish reduces the stock and increases the cost of fishing so that resource rents dissipate (and therefore $\mu = 0$). This competition leads to excessive fishing effort (too many vessels and too much labor) and drives stocks to levels that are too low from society's point of view (Fig. 2b).

The optimization of economic surpluses requires the implementation of appropriate regulations (often referred to as "conservation and management measures") in the fishery. Many different types of fishery regulations have been devised, but only a few have the potential to optimize economic surpluses. Economically optimal measures impute a price to the fish, through either the imposition of taxes on harvests or on
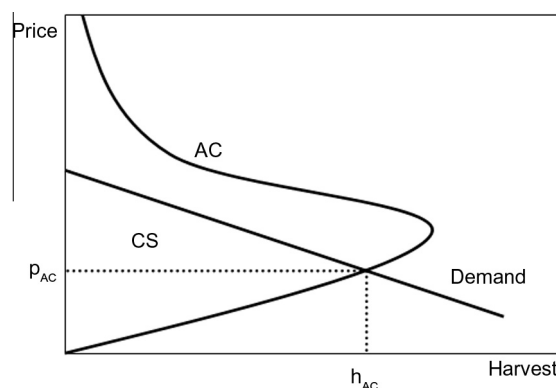
Figure 2b *Open-access* fishery with price equal to average cost (AC). The market equilibrium occurs at price, p_{AC} and harvest, h_{AC}. Average cost comprises a backward-bending supply curve. There is no producer surplus, and therefore no resource rent, and only consumer surplus (CS) is realized.

fishing effort or the assignment of transferable rights in shares of allowable catch (quota rights) or in fishing areas.

Commercially exploited fish stocks depend upon the ocean habitat (or ecosystem), O, where they are found, including wetlands, coral reefs, the seafloor, seagrass beds, water quality, ecological relationships, and sources of food. Analogous to the "black box" representing the economic technology for producing seafood from an array of inputs, we define a "blue box" to represent the ecological "technology" for producing fish in the sea from components comprising ocean habitat (Fig. 1). Indeed, ocean habitat is a *sine qua non* of the fishery, and the carrying capacity, K, of the environment may be viewed as a positive function of habitat, possibly with diminishing returns: $\{K(O): K' > 0, K'' \leqslant 0\}$. Further, because the growth of the fish stock depends upon the carrying capacity of the ecosystem, growth could be modeled explicitly as a function of ocean habitat as well: $F(X,O)$.

3. Ocean habitat and ecosystem services

Because ocean habitat supports commercial fish stocks, ocean habitat can be conceived as another type of input in the production of seafood. Under this interpretation, ocean habitat provides what is often referred to as an "ecosystem service." In theory, ocean habitat can be partitioned into its component parts, and each of these components can be interpreted to provide an ecosystem service. Typically, there is much uncertainty about the blue box technology, *i.e.*, how these components interact to produce the composite ocean habitat. Further, there may be either natural or human-induced variations in the components. Finally, there may be inexact measures of the ecological contribution of each of these components (*i.e.*, early studies modeled wetland acreage as the only relevant proxy for habitat, and even some modern studies find a similar result (Aburto-Oropeza et al., 2008)).

In theory, the economic surpluses that result from the commercial harvesting of fish can be imputed to the ocean habitat in aggregate (or to its components individually), thereby assigning an economic value to the ocean habitat as an ecosystem service. This method of valuing the ecosystem services of

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[3] Fishing effort here is a composite variable representing all of the inputs (other than fish) in the harvest of fish.

ocean habitat is called the "productivity approach".[4] It is appropriate to impute both producer and consumer surpluses in the downstream seafood markets to measure the value of ecosystem services. In practical applications, however, because of data limitations or ecological uncertainties, often only a portion of surpluses, such as resource rents, are used as a measure of value. Finally, value can be imputed either to the fish stock, the ocean habitat supporting the fish stock, or the individual components comprising ocean habitat. Assigning value to more than one of these three types of services simultaneously would lead to a double-counting of values.

The productivity approach is a supply-side method for valuing ecosystem services (Ellis and Fisher, 1987). It should be contrasted with other valuation approaches that focus on the willingness-to-pay (WTP) for the non-marketed attributes of the ocean ecosystem.[5] WTP approaches include the travel cost, hedonic pricing, and contingent valuation methods of estimating the benefits to individuals or firms. In situations where a total economic value (TEV) is being estimated, it may be appropriate to apply multiple valuation methods, such as combining the productivity and WTP approaches.[6] DeGroot et al. (2012) review and summarize recent estimates of TEVs for ecosystem components, including those for open oceans, coral reefs, coastal systems, and coastal wetlands, emphasizing the uncertain and contextual nature of the estimates.

In developing a productivity measure of the value of ocean ecosystem services, it is important to include all of the fisheries that are supported by the ocean habitat. The development of such a comprehensive productivity measure raises questions about the optimal mix of fisheries, where gear conflicts, bycatch, or other external costs may arise among disparate fishery sectors or fleets.

In the prototypical case in which market price is unaffected by supply from the fishery, the use of supply-side measures to value the ecosystem services provided by ocean habitat depends crucially upon whether fish are managed very well. If the fishery is unregulated, and the fish are thereby overexploited, the value imputed to the habitat would be diminished considerably. In such a case, economic surpluses would be limited to those accruing only to the more skilled fishermen (so-called "highliner" rents). Where fishermen are similar in terms of skills, technology, and behavior, theory suggests that resource rents would be dissipated completely. Consequently, the imputed value of the habitat would be close to zero (cf., Lynne et al., 1981). Where a fishery is regulated to maximize the net social benefit, the value of the ocean habitat supporting the fishery is also thereby maximized.

The ocean can be of varying quality for fishery habitat, depending upon its natural characteristics, the nature of

ecological relationships, and the extent of human influences. Although it is obvious that the characteristics of habitat provide support for fish stocks, in many cases the physical, chemical, and ecological relationships between the habitat and the growth and maintenance of fish stocks are poorly understood. Consequently, it can be problematic to understand how changes in habitat lead to changes in fish stocks (Armstrong and Falk-Peterson, 2008).

In many applications, stakeholders and decision-makers are interested in estimating the economic damages associated with changes to habitat. For example, the loss of wetlands due to coastal development could lead to changes in the stock size of fisheries that depend upon the wetlands as nurseries, for protection from predators, or as a source of food. In these situations, it is critically important to understand the physical, chemical, and ecological relationships. Further, there may be value to undertaking scientific research that uncovers the nature of these relationships (Barbier et al., 2008).

4. Consumer surplus in seafood markets and distributional effects

Where the demand for fish is inelastic, it is important to consider the distributional effects of management measures. Freeman (1991) has shown that there could be a larger surplus gain from the improvement of habitat in an open-access fishery than in a well-managed fishery. In seafood markets where price is inelastic, an increase in habitat shifts both average and marginal costs down. In the open-access case, average cost provides the relevant supply schedule. There are no resource rent gains in this case, because rents have been dissipated completely. Consequently, only consumer surplus gains are realized as price decreases when average cost shifts down (Fig. 3a). On the other hand, in the well-managed fishery, marginal cost is the relevant supply schedule. There are surplus gains to both producers and consumers (a portion of consumer surplus gains actually are a transfer from producers and

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[4] This methodology is also referred to as the "production function approach"(Mäler, 1991; Barbier, 1994), "valuing the environment as an input" (Ellis and Fisher, 1987), or a "natural production function" (Bell, 1998).

[5] Another valuation approach is referred to as the replacement cost approach. This approach measures the cost of a substitute for a natural function, such as the costs of water treatments, wetland restoration, or the construction of artificial habitat (Milon, 1989).
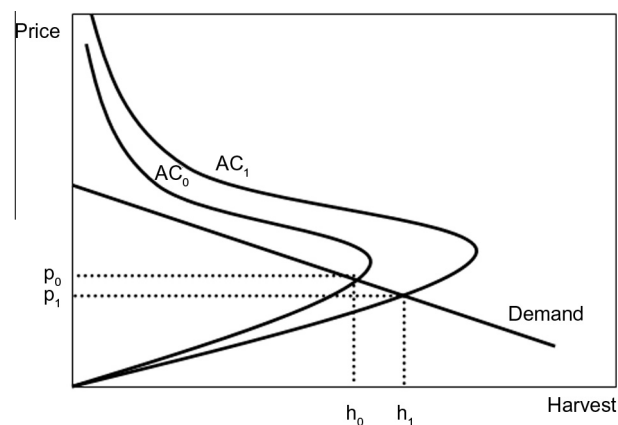
[6] Further, a complete measure, also known as total economic value (TEV), would take into consideration the value of the ocean as an input into all other human uses of the ocean, such as for waste disposal, for shipping, for recreation, or as an aesthetic resource (e.g., ocean views).

Figure 3a Open-access fishery with price equal to average cost, as in Fig. 2b. The initial market equilibrium occurs at price, p_0 and harvest, h_0. Improvements in ocean habitat lead to an expansion of the target fish biomass, reflected in a shift of average cost to the right, from AC_0 to AC_1. The new market equilibrium occurs at p_1 and h_1, where yields are greater and price is lower. Even with these shifts, which could lead to a larger consumer surplus, there is no resource rent.
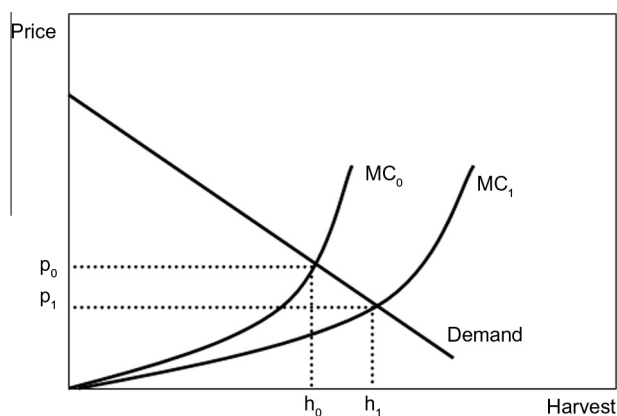
Figure 3b *Well-managed* (rationalized) fishery with price equal to marginal cost (MC), as in Fig. 2a. The initial market equilibrium occurs at price, p_0 and harvest, h_0. Improvements in ocean habitat lead to an expansion of the target fish biomass, reflected in a shift of marginal cost to the right, from MC_0 to MC_1. The new market equilibrium occurs at p_1 and h_1, where yields are greater and price is lower. At the new equilibrium, there are increases in combined consumer and producer surpluses, resource rents are realized, and there is a partial transfer of surpluses from producers to consumers.

Figure 4 An *open-access* fishery with an expansion of demand due to an increase in seafood quality in combination with improvements in ocean habitat. (Note that higher seafood quality could be a consequence of the habitat improvements.) The initial market equilibrium occurs at price, p_0 and harvest, h_0. Demand expands from D_0 to D_1 because of higher seafood quality, and average cost shifts to the right, from AC_0 to AC_1, due to habitat improvements. The new market equilibrium occurs at p_1 and h_1, where yields are much lower and price is much higher. Consumer surpluses (the only source of value in an open-access fishery), reflected after the shifts by the area below D_1 and above p_1, has declined significantly as a consequence of environmental improvements.
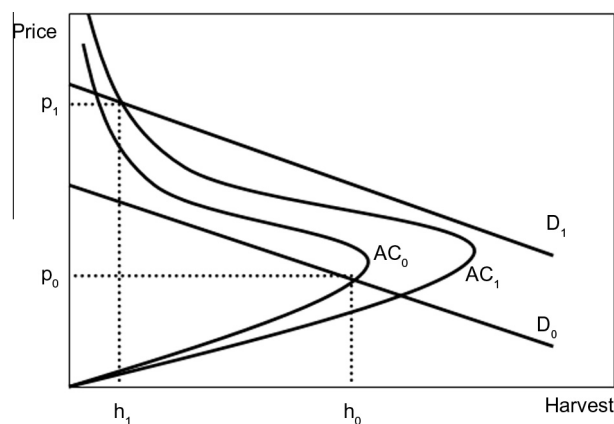
therefore do not count as a social gain) (Fig. 3b). Freeman shows that it is theoretically possible that the gains to consumer surplus in the open-access fishery exceed the combined gains to producer and consumer surpluses in the well-managed fishery.

Thus, in the case of a commodity where demand is relatively unresponsive to price changes (*i.e.*, there may be few close substitutes), there may be significant gains from improvements in habitat. As a corollary to this finding, there may be larger losses (again to consumer surplus) from the degradation of habitat in the open-access case with inelastic demand. Consequently, in such a situation, a policy to manage fisheries to maximize economic yield should lead to smaller losses relative to open-access when faced with impending or ongoing habitat degradation.

McConnell and Strand (1989) consider a different ("demand side") case where seafood demand is affected by water quality. They show that, in the open-access case, where consumer surplus is the only relevant measure of welfare, improvements in water quality in theory could expand demand to the point where the fishery is seriously overexploited, raising price and lowering harvests. In the extreme, improvements in water quality could actually decrease welfare (Fig. 4). The corollary to this finding suggests that the loss of ocean habitat, as measured by the degradation of water quality, could lower seafood quality, thereby contracting demand, and improve welfare as price drops and harvests increase.

5. Valuing ecosystem services

Some of the earliest efforts at valuing ecosystem services are now widely regarded as methodologically inadequate or even invalid. Among these were efforts to estimate the energy flow through wetlands using an energy theory of value (Gosselink

et al., 1974).[7] Nevertheless, these early efforts served an important purpose in motivating economic research on the productivity approach. In this section, we examine three supply-side approaches to valuing ecosystem services, and, in the next section, we present an example of the use of these approaches for the traditional fisheries of the Red Sea. To enhance comparisons, all value estimates in this paper have been adjusted to 2012 US dollars, using an exchange rate of *SAR*3.75/$1.00 and deflating with the US Consumer Price Index.

5.1. The total value approach

An early effort to value ocean habitat was sponsored in the early 1970s by the Center for Wetland Resources at Louisiana State University (LSU) (Pope and Gosselink, 1973; Gosselink et al., 1974). LSU researchers developed estimates of the value of tidal wetlands using estimates of: (i) wild harvest shellfish sales and value-added per acre; (ii) expenditures for saltwater fishing, hunting, and boating per acre; (iii) potential cultured

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[7] Gosselink et al. (1974) attempt to value the energy flow through a marsh as represented by estimates of primary productivity. The authors use a controversial "life support" approach to value energy flows, using a price of energy obtained by dividing GNP by domestic energy consumption. The application of this approach yields estimates on the same order of magnitude as those developed by examining sales, expenditures, and avoided costs of waste treatment. The life-support approach has been criticized by economists as an inappropriate "energy theory of value," in which energy content is incapable of reflecting relative scarcity the way prices do in a market economy (Shabman and Batie, 1978; Huettner, 1976). Intuitively, the "price" of energy calculated using this approach cannot adjust to reflect surplus or scarcity relative to other goods and services precisely because it is fixed.

shellfish sales and value-added per acre; and (iv) the avoided costs of secondary and tertiary treatment of domestic waste and the avoided costs of phosphorous removal from domestic waste. Combining estimates of commercial and sport fisheries, intensive oyster culture, and tertiary treatment, these authors estimate a value of $18,875 per acre of wetlands. Food production, in the form of wild harvest and cultured shellfish, represent about 40 percent of the total, or $7550 per acre.

A second effort to value ocean habitat was subsequently published by two scientists at the US Environmental Protection Agency (Tihansky and Meade, 1976).[8] These authors estimated the value of US estuaries by calculating estuary acreage and dividing it into the exvessel value (the value of—typically unprocessed or minimally processed—seafood landed at the dock) of those fish and shellfish known to be dependent upon estuaries. Importantly, they attempt to take into consideration all of the fisheries and shellfisheries that depend upon the estuarine type of ocean habitat. In this approach, the sales value of those fisheries dependent upon estuaries (estimated to be about $4.2 billion for the United States in 1967) embodies the value of the estuaries. For US estuaries, the authors estimate a capitalized value[9] of $4920 per acre (2012 dollars). Using this method, the authors examine the relative productivity of estuaries in alternative locations, for example, finding per acre values of $58,000 for San Francisco Bay; $41,200 for Delaware Bay; $26,000 for Penobscot Bay; $16,000 for Tampa Bay; and $7800 for Narragansett Bay (all estimates are expressed in 2012 dollars).

These publications were criticized heavily by economists, who were quick to point out conceptually more appropriate methods from the field of economics (Shabman and Batie, 1978; Batie and Shabman, 1982). In particular, the following needs were identified:

1. Economic surpluses, not commercial sales, are the relevant measure of economic value.
2. The size of fish stocks should be linked explicitly to the relevant habitat, while holding the effects of fishing effort on stock size constant. In other words, it is important to understand the relationship between the geographic measure of the habitat and its ecosystem services. Further, fishing effort is a partial substitute for habitat in the production of fish.
3. The focus should be on the effects of *marginal changes* in the value of habitat (Batie and Wilson, 1978), not the value of the entire habitat.[10] (Note that it is unlikely that all tidal wetlands would be lost to development, and, even if that were to be the case, some fisheries would continue to be viable.)
4. Variations in the productivity of habitat in different locations should be considered.

5. The cost of putting in place potential habitat substitutes (e.g., artificial reefs, restored wetlands, and restocking programs) might be considered as an alternative to using economic surpluses as a measure of value. (Note that the substitutes must provide similar levels and qualities of ecosystem services, and they must cost less than any foregone surpluses.)

Even with these criticisms, the total value approach continues to be used, as it is one of the simplest methodologies, representing basically a way of standardizing data on the value of fish harvests. For example, White et al. (2000) use the approach to estimate the economic losses associated with coral reef destruction in the Philippines. The authors multiply the average price of sustainable fisheries for local consumption times a production range to develop an estimate of the value range for coral reefs of $24,000/km^2 to 70,000/km^2 (2012 dollars).

5.2. The marginal productivity approach

A number of authors have taken an approach based on production theory to evaluate the marginal productivity of ocean habitat in steady-state (Aburto-Oropeza et al., 2008; McArthur and Boland, 2006; Barbier and Strand, 1998; Bell, 1989; Lynne et al., 1981; Batie and Wilson, 1978). The marginal productivity approach is practical in the sense that it does not assume that a fishery is being managed optimally. (As a consequence, the imputed values for ocean habitat sometimes can be quite small.)

An obvious initial choice for a production function is the Cobb–Douglas form, and several authors have modeled the value of ocean habitat using it (Bell, 1997; Batie and Wilson, 1978). The harvest of fish in period t is written as:

$$h_t = \gamma E_t^\alpha O_t^\beta,$$

where the sum $\alpha + \beta$ determines the type of returns to scale, and γ is a (catchability) constant. The Cobb–Douglas form is well-known to economists, but it does not rely explicitly upon renewable resource theory. It may be appropriate for analyzing data that lie within a narrow range.

An alternative specification is to estimate the parameters of a production function in which the biological growth of a fish stock has been taken into account (Lynne et al., 1981). In the literature, this is commonly referred to as a Gordon-Schaefer production function. To start, a model of the relationship between ocean habitat and the productivity of the fish stock is hypothesized. This relationship typically takes the ecological carrying capacity, K_t, to vary in each period as a function of some measure of ocean habitat, O_t:[11]

$$K_t = f(O_t)$$

A typical Schaefer production function for the harvest, h_t, of fish in any period is written as:

$$h_t = qE_tX_t(f(O_t)),$$

where E_t is a composite measure of fishing effort. The fish stock is assumed to grow logistically:

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[8] These authors refer to even earlier effort by ecologists at the University of Georgia to value estuaries and salt marshes using a similar methodology.

[9] The authors use a discount rate of 10% to capitalize annual gross revenues per acre of estuary; here, we re-express their estimates using a discount rate of 5%.

[10] The same issue resurfaced in a more recent attempt to value the world's ecosystem services and natural capital (Costanza et al., 1997), and it received the same criticism. In the Costanza et al. (1997) study, estuaries receive an average annual value of $14,245 per acre (adjusted to 2012 dollars). Food production represents only about two percent of this figure.

[11] Many of the relevant studies examine wetland acreage as the measure of habitat, and this function typically is specified simply as $K_t = \alpha \cdot \ln O_t$, where α is a parameter to be estimated.

$$\frac{\partial X_t}{\partial t} = rX_t(1 - \frac{X_t}{f(O_t)})$$

At steady-state, fish harvest equals stock growth, and harvest can be written as a function of ocean habitat and fishing effort:

$$h_t = qf(O_t)E_t - \frac{q^2}{r}(f(O_t)E_t^2)$$

Assuming a functional form for $f(\cdot)$, the parameters of this equation can be estimated, using data on fish catch, measures of ocean habitat, such as coral reef, seagrass or mangrove acreage, and fishing effort.

Where the price of seafood is p_t, the marginal value product of ocean habitat can be calculated as:

$$\text{MVP}_O = p_t\frac{\partial h_t}{\partial O_t}$$

MVP_O can be interpreted as the price of the ecosystem services of the ocean habitat in the support of fish stocks. Table 1 presents some alternative estimates of MVP_O from a number of authors, showing the wide range of estimates of the value of ocean habitat.

There are a number of variations to the marginal productivity approach that have been introduced. Employing a Gordon–Schaefer production function for valuing the productivity of salt marshes in yields of blue crabs, Lynne et al. (1981) include harvest in the previous period, h_{t-1}, as an additional explanatory variable. This variable in effect represents a distributed lag of the effects of ocean habitat and fishing effort in the previous period (Bell, 1989). The addition of this variable improves model fit slightly, but increases the standard errors of the estimated parameter values.

Employing a Cobb–Douglas production function to model oyster yields in Virginia, Batie and Wilson (1978) examine the marginal productivities of wetland acres and salinity as proxies for habitat. The authors find only that fishing effort and salinity are significant predictors of oyster yields. Notably, the dataset is limited to cross-sectional data for Virginia counties in only one year.

Barbier and Strand (1998) employ a Gordon–Schaefer production function that incorporates an unusual formulation for the fishery growth function. The authors examine the marginal productivity of mangrove wetlands for the shrimp fishery in Campeche, Mexico. They modify the logistic function as follows:

$$\frac{\partial X_t}{\partial t} = rX_t(f(O_t) - X_t)$$

This equation behaves in the same manner as the more typical Schaefer formulation, although, for identical parameter values, its effect is to amplify the effects of intrinsic growth and the size of the stock.[12] At steady-state, the harvest function is written as:

$$h_t = qf(O_t)E_t - \frac{q^2}{r}E^2$$

Because the carrying capacity as a function of ocean habitat does not appear in the second term on the right-hand side, the marginal value product is affected only positively by ocean habitat. As a consequence, the imputed value of wetland habitat is much larger than that estimated using the more conventional logistic form.

The marginal productivity approach uses a constant price and makes no assumptions about the presence of resource rents. This approach can be extended to the cases of inelastic demand, expansion of demand, and optimal regulation, or all three. Using water quality as a measure of ocean habitat, McConnell and Strand (1989) extend the marginal productivity approach by showing the change in social returns for an improvement in water quality. In their presentation, an inelastic demand shifts out because increases in water quality lead to higher quality seafood. Moreover, marginal cost shifts down, increasing the resource rent, because of the increased productivity of the habitat. The net benefits are calculated as changes in the relevant areas between old and new demands and old and new marginal costs.

Barbier (2003) employs a similar approach to evaluate changes in mangrove loss for fisheries exhibiting inelastic demand in Thailand. Here, because of the open-access nature of the Thai fisheries, changes in consumer surplus only are the relevant measures of the value of changes in mangroves as habitat. The author measures the welfare impacts at steady state (Table 1), although the possibility of demonstrating the welfare impacts of changes in habitat on the approach to steady-state is suggested in theory.

5.3. The residual rent approach

The fundamental bioeconomic model, developed first by Gordon (1954) and Scott (1955) and elaborated by Clark (1990), Anderson (1986), and others (Flaaten 2011) can be viewed as the basic way of valuing ocean habitat by proxy. (Essentially, in this approach, the fish stock, X, comprises the ecosystem service.) The value of the resource rent, μ, in an optimally managed fishery facing a constant price constitutes the value of the ecosystem services that support the fish stock. This approach also has been described as the "residual rent approach".[13]

In the simplest models, the fishery supplies only a small part of the total market for the fish as seafood, so market price is treated as a constant, and consumer surplus is not accounted for as a component of the value of ocean habitat. Ocean habitat provides support for all commercial fish species, and therefore it should be valued as the sum of resource rents from the various fisheries that it supports. In these models, estimates of the costs of fishing in one or more periods are subtracted from gross revenues to derive residual rents.

Only a few studies, many not easily accessible, have estimated the ecosystem service value of fisheries dependent upon on coral reef habitat using the residual rent approach (see the sources in deGroot et al., 2012). Many of these studies utilized very crude approaches, which are not based on bioeconomic models and which sometimes omit explicit calculations, to estimate residual

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[12] This formulation may make sense for an annual crop species such as shrimp.

[13] Strictly speaking, the resource rent accounts for the *dynamic* stock effects in a fishery, whereas the residual rent is a measure of *static* revenue minus cost. Tihansky and Meade (1976) first suggested calculating the resource rent as a "residual" when all of the costs of fishing effort are subtracted from total revenues. Because resource rent is a function of the state of the fishery, this residual rent method has been utilized rarely (but see Lange (1993) for an application to green accounting). In theory, open-access fisheries dissipate resource rents, yielding no serviceable estimate of residual rent for ocean habitat. The open-access nature of many fisheries calls for a more pragmatic approach to valuation.

Table 1 Comparisons from the literature showing a wide range of estimates of marginal value products (MVP$_O$) for different categories of ocean habitat (all estimates expressed in the log of 2012 US dollars).

Authors	Location	Ocean habitat	Fish stock	ES function	Marginal value product [log $(2012)/km^2]
Lynne et al. (1981)	Florida Gulf Coast, United States	Salt marsh	Blue crab	Semilog	2.5
McArthur and Boland (2006)	South Australia, Australia	Seagrasses	Calamary Blue crab Garfish King George whiting Snapper Tommy ruff Yellowfin whiting	Log–Log	3.6
Bell (1989)	Florida East and Gulf Coasts, United States	Wetland	Blue crab Stone crab Spiny lobster Red snapper Grouper Oyster Black mullet Shrimp	Semilog	4.3
Barbier (2003)	Gulf of Thailand and Andaman Sea, Thailand	Mangroves	Shrimp Crab Squid Cuttlefish Demersal finfish	Semilog	4.6
Barbier (2007)	Gulf of Thailand and Andaman Sea, Thailand (dynamic estimate from Thai data)	Mangroves	Shrimp Crab Squid Cuttlefish Demersal finfish	Semilog	4.7
Batie and Wilson (1978)	Virginia	Wetland	Oyster	Linear	4.8
Barbier and Strand (1998)	Laguna de Terminos, Campeche State, Mexico	Mangroves	Shrimp	Linear	5.8
Aburto-Oropeza et al. (2008)	Gulf of California, Mexico	Mangroves	Finfish Crab	Square root	6.7

rents. Berg et al. (1998) calculated estimated potential revenues at maximum sustainable yield net of operational and labor costs to impute an estimate from commercial fisheries of $12–15,000/km^2 for Sri Lankan coral reefs. In contrast, with only minimal information, Samonte-Tan et al. (2007) suggested that marine fisheries in the Bohol Marine Triangle of the Philippines likely yield residual rents attributable to the coral reef habitat on the order of $50–200,000/km^2.

Questions about the relative contributions of each of the components of ocean habitat to the value of fish are more problematic and remain largely unexplored. Sanchirico and Mumby (2009) developed a theoretical model to examine mangroves and seagrass beds as substitutes in the ecological production of fish that recruit to a coral reef fishery. Entry limitations are imposed on the fishing fleet, and rents are approximated with a license fee. Service values for mangrove (or seagrass bed) acreage can be measured as the marginal change in the combined total values of profits and license fee payments with a marginal change in acreage.

6. An example: the traditional fisheries of the Red Sea

Jin et al. (2012) apply standard bioeconomic methods to characterize the current status and the potential of the traditional fisheries off the northern and central regions of the Red Sea coast of the Kingdom of Saudi Arabia (KSA). (These two regions can be distinguished ecologically from a southern region, for which statistics are harder to compile.) In these mainly artisanal coastal fisheries, fishermen deploy hand-lines, gillnets, and traps from small, open motorboats. They catch and land groupers, snappers, emperors, barracudas, scads/jacks/trevallies, kingfish, and tunas. These species are known to spend significant portions of their lifecycles in association with the coral reefs along the coast, and thus estimates of the value of the fisheries could be used to impute values to the services provided by the coral reef system.[14]

Using time-series data on fishing effort (boat-days) and landings obtained from the KSA Department of Marine Fisheries, the authors apply a Clarke–Yoshimoto–Pooley approach to estimate the collective intrinsic growth rate, r, carrying capacity, K, and catchability, q, for these seven stocks. With these parameters and with calculations of a weighted average

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[14] We thank one of the reviewers for pointing out that, to the extent that some of these species may spend significant portions of their lifecycles in ocean areas outside of coral reef systems, we may be overstating the traditional fisheries ecosystem service values for the reef systems *per se*. Refinement of our estimate would require ecological studies of the extent of the reliance of these species on local Red Sea reef systems, and we identify this as an important issue for future research.

Table 2 Annual per-unit ecosystem service values for the Red Sea coral reefs.

Fishery status	Biomass (mt)	Effort (b-d)	Economic "Values" ($ millions)		ES valuation methods ($000/km^2)		
			Gross revenue	Resource rent	Total value	Marginal productivity	Residual rent
OY	6439	367,000	46	0	9.25	1.23	0.00
MSY	13,850	208,000	56	30	11.25	1.49	6.23
MEY	18,644	146,000	53	35	10.65	1.41	6.97

Abbreviations: b-d = boat-days; ES = ecosystem service; km = kilometer; MEY = maximum economic yield; MSY = maximum sustainable yields; OY = optimal yield; mt = metric tons.

price across the seven target species and of fishing costs, the authors calculated actual open-access (OA) yields and theoretical maximum sustainable and economic yields (MSY and MEY).

The authors found that current annual fishing effort levels, ranging between 300,000 and 350,000 boat-days, constituted clear over-fishing of the stocks. Exploitation at these levels resulted in combined OA yields of 7725 mt, leaving standing stocks of 6440 mt. Total revenues were estimated to be $46 million, but net social benefits were estimated to be zero. In contrast, using a 5% rate of discount, the authors calculated a socially optimal level of fishing effort at 146,000 boat-days. Optimal exploitation at this much reduced level would result in MEY yields of 8901 mt with standing stocks of 18,644 mt. Total revenues at MEY were estimated to be $53 million, with net social benefits of $35 million. The authors also calculated yields, standing stocks, revenues, and resource rents for MSY (Table 2).

Using these estimates, we can approximate the ecosystem service values of the coral reef-seagrass bed systems (Table 2).[15] Ideally, one would like to model the blue-box technology from Fig. 1 directly, given that many factors likely contribute to the biological productivity of the Red Sea fish stocks, including water quality (temperature, salinity, transparency, and nutrient levels), mangrove cover, seagrass beds, and coral reefs, among others. Further, these factors may vary geographically and over time in terms of their relative contributions to fishery productivity. Without detailed information on the blue-box technology and its spatial and temporal variability, for expositional purposes, we assume that the most important ecosystem service is a homogeneous and unchanging coral reef-seagrass complex. For the northern and central regions combined, the area of this complex has been estimated to be 5000 km^2.[16]

The calculations of annual per-unit ecosystem service values are straightforward. Using the total value approach, we divide gross revenues by the area of the coral reef-seagrass bed complex to obtain the annual per-unit value. We do the same thing with estimated resource rents to obtain the annual per-unit value for the residual rent approach. The marginal pro-

ductivity approach requires an assumption about the relationship between yields and ocean habitat. Assume that carrying capacity is directly proportional (here equal) to ocean habitat, $K{\sim}O$. Jin et al. (2012) apply the following Fox-type production function to the fishery:

$$h = qEKe^{-q/rE}$$

Letting $K = O$, and taking the derivative with respect to O gives:

$$\frac{dh}{dO} = qEe^{-q/rE}$$

and the marginal productivity measure is calculated as:

$$\text{MVP}_O = p\frac{dh}{dO} = pqEe^{-q/rE}$$

Table 2 presents estimates of the imputed per-unit ecosystem service values ($/km^2) for the reef-seagrass complex using each of the three methods described above. The ecosystem service values obtained using the "total value" approach are much larger than those obtained with the two other methods. We have discussed already the reasons why these estimates may be inappropriate. Note that with both the total value and the marginal productivity approaches, the MSY estimate exceeds both the OA and MEY estimates. This result is due to the nature of the underlying biological production, ensuring that MSY leads also to the maximum total revenues and marginal values. Under either approach, whether the ecosystem service values at MEY exceed or fall below those at OA depends critically upon the costs of fishing, so that high costs could lead to the (seemingly counter-intuitive) result that OA ecosystem service values exceed those at MEY.

Only when employing the residual rent approach do we obtain the sensible result that ecosystem service values are maximized when the fishery is managed rationally to optimize resource rent. Resource rent is zero under OA. Some level of

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[15] In Table 2, we compare the estimated annual per-unit ecosystem service values at three equilibrium points of the bio-economic model for a fixed carrying capacity. In general, these per-unit service values may change (non-linearly) with respect to the carrying capacity, which is affected by changes in the quality or quantity of the relevant ocean habitat. See Kahn and Kemp (1985) for a descriptive derivation of such changes in the case of submerged aquatic vegetation in Chesapeake Bay.

[16] Unpublished estimate developed by graduate students enrolled in the Marine Science Program at the King Abdullah University of Science and Technology and working with one of the current article's co-authors (Kite-Powell).

Table 3 Capitalized (at 5%) per-unit ecosystem service values for Red Sea coral reefs.

Fishery status	ES valuation methods ($000/km^2; Capitalized @ 5%)		
	Total value	Marginal productivity	Residual rent
OY	184.9	24.6	0.0
MSY	225.1	29.9	124.5
MEY	213.1	28.3	139.5

Abbreviations: b-d = boat-days; ES = ecosystem service; km = kilometer; MEY = maximum economic yield; MSY = maximum sustainable yields; OY = optimal yield; mt = metric tons.

resource rent is realized at MSY, but this is exceeded by that obtained at MEY. Importantly, MEY leads to an outcome where there is a larger standing stock. Although the existence of higher stock levels has not been explored here, there may be ancillary benefits to recreational fisheries, ecosystem health (say, through the grazing by fish of algae on coral reefs), and insurance against natural or anthropogenic hazards (storms, harmful algal blooms, and pollution events).

In this example, we have assumed that the full resource rent should be imputed to the coral reef-seagrass bed complex; in reality, we would want to specify a blue-box ecological production function, distributing the rent across relevant ecosystem factors. When used in a cost-benefit context, we also should consider capitalized per-unit ecosystem service values (Table 3). Under MEY, capitalized ecosystem values derived from the traditional Red Sea fisheries and summed across all of the relevant coral reefs are on the order of $1 billion, surely rivaling the net benefits of some forms of industrial development.[17] Finally, note that we have estimated ecosystem service values for the traditional fisheries only (a subset of "provisioning services"). There are likely to be significant service values contributing to the industrial and recreational fisheries, recreational diving, passive non-market benefits, and other valued endpoints. While extant estimates of an array of other sources of value, including provisioning, regulating, habitat, and cultural values, are known to be uncertain and dependent upon context, a recent compilation of estimates suggests that provisioning services (interpreted broadly) may account for only about 15% of the ecosystem service values for coral reefs (deGroot et al., 2012).

7. Ecosystem-based management considerations

Our focus has been on ocean habitat, *per se*. Fishery resources exist within an ecosystem context, and species interactions may be important in determining the productivity of fisheries. In particular, other species, guilds, or trophic levels may be understood to provide ecosystem services, and a full treatment of ecosystem services may require the characterization of ecological interactions. Further, coastal development can be conceptualized as a process of exploiting a non-renewable resource with implications for the health of an associated renewable resource. These interactions fall within the realm of ecosystem-based management.

7.1. The multispecies approach

In cases where species interact ecologically, the resource rent from one or more commercial species could be used to impute a value to species that are not utilized commercially.[18] For example, high trophic level commercial fish species may be predators of lower trophic level species. Consequently, it is possible to conceptualize ocean habitat as embodied in the lower trophic level prey. Where the latter are not commercially exploited, their value may be imputed using resource rents from the commercial fishery. Ragozin and Brown (1985) develop a model of a predator–prey system in which the economic value of the prey is estimated as a function of the resource rents realizable from the harvest of the predator. According to the system, a predator stock, X, is affected positively by the size of the prey stock, Y, and the prey stock is affected negatively by the size of the predator stock. Let $F(X,G(Y))$ and $G(Y,F(X))$ be the growth functions for the predator and prey, respectively. The authors find that the steady-state current value resource rent for the prey is defined as:

$$\mu_Y = \mu_Y G_Y + \mu_X F_X$$

where the value of ocean habitat comprising the prey is μ_Y. This value is just matched by the sum of marginal productivities from both stocks, weighted by their marginal values. The interpretation of this finding is that the value of the prey is reflected in future growth values for both the prey stock, $\mu_Y G_Y$, as food for the predator, and the predator stock, $\mu_X F_X$, which depends upon the prey.[19]

Even if it is not possible to model the blue box technology in full–or even in part–welfare gains seem likely from improvements in fishery management, and the resulting rents could be used as a broad proxy for the services of ocean habitat. In theoretical work on multispecies population dynamics on Caribbean coral reefs involving a generalist predator and two prey fish species, Kellner et al. (2011) find that the largest welfare gains may obtain from improved fishery management–and not from improvements in understanding ecological interactions.

7.2. Interdependent renewable and nonrenewable resources

Most of the literature on the productivity approach assumes that a steady-state exists in harvests and fish stocks, and the value of ocean habitat is estimated at the steady-state. Swallow (1990) models coastal development as the extraction of an exhaustible resource stock that also provides habitat for a fishery. In this formulation, the social optimum must consider the value of both coastal development and the fishery. Resource rent to the fishery can be written as follows, where the growth of the fish stock $F(\cdot)$ depends upon the exhaustible resource O:

$$\mu = \frac{c_X F(X,O)}{F_X(X,O) - \delta}$$

Unlike the earlier formulations, ocean habitat itself is diminishing at a rate that depends upon the rate of coastal development. As a consequence, the efficient harvest level in each period, expecting smaller stocks in the future, exceeds harvests that would occur when coastal development is unchanging. The diminishing ocean habitat implies that resource rents to the fishery also are smaller than they would

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[17] As discussed in an earlier section, this estimate of the ecosystem service value for the entire reef would be relevant only if the entire reef is threatened by coastal development. Further, the estimate ignores differences in reef quality, particularly the likely lower values arising from already degraded reefs.

[18] Allen and Loomis (2006) employ a similar approach to impute the results of nonmarket contingent values for species at high trophic levels to develop estimates of partial willingness-to-pay for species at lower trophic levels.

[19] This description is an oversimplification, as the steady-state resource rent, μ_X, is itself a positive function of the prey's resource rent. Notably the predator's rent also depends positively upon the constant price, p, of the predator, making the prey's resource rent indirectly a positive function of p as well.

be in the steady-state when habitat is unchanging. Therefore, according to Swallow, estimates of the imputed value of ocean habitat to the fishery are overstated when a steady-state is assumed. Note that, in this case, additional rents are generated from the exhaustible resource production. When multiple industry sectors are involved, the resource valuation can become even more complex.

8. Discussion

Commercial fisheries are just one of many human activities that use ocean habitat in the Red Sea. Red Sea habitat also may be valued passively for its existence, for the use of future generations, or in order to leave options open for future uses. Some human activities may diminish the value of ocean habitat for other activities or passive uses. These include the over-exploitation of fish stocks, the pollution of the marine environment through the release of nutrients or the spilling of hazardous materials, and the clearing of mangroves. Some human impacts may be irreversible, such as construction activities in mangrove wetlands or the sedimentation of coral reefs from coastal erosion near developments.

It is important to estimate the value of ocean habitat in order to make rational decisions about the best mix or sequence of coastal and ocean uses. A supply-side approach values ocean habitat using estimates of consumer and producer surplus in downstream markets. In essence, ocean habitat is treated as an input (alternatively, it provides an "ecosystem service") in the production of commercial goods and services.

We examine supply-side valuation approaches for ocean habitat using commercial fisheries as a relevant example. Similar techniques may be employed to value ocean habitat for subsistence and recreational fisheries. Other methods may be needed for valuing passive uses. In the case of commercial fisheries, typical supply-side valuation involves modeling ecological carrying capacity for fisheries as a function of ocean habitat. Extant studies consider one measure of ocean habitat, such as wetland acreage, the proportion of the seafloor covered by submerged aquatic vegetation, or a water quality index. The most appropriate methodological approach depends upon the availability of data as well as the institutional setting. Because non-market valuation methods also require accurate information on the relevant ecological and economic interactions, the results of supply-side valuation studies can provide insights beneficial to these studies as well.

In order to estimate the value of ocean habitat, data must be compiled on fish landings, fishing effort, seafood prices, and one or more measures of ocean habitat. Statistical estimation requires variation in all of these variables, including the measures of ocean habitat. This variation might be observed in either cross-sectional data, time-series data, or both.

Institutional settings–both market and regulatory–are important. We expect that resource rents will be small or non-existent in open-access fisheries. Consequently, consumer surplus may be an important measure of economic value. Consumer surplus may be affected when demand for seafood is inelastic or when seafood quality depends upon the condition of ocean habitat (e.g., water quality). Where demand is inelastic, due to the absence of close substitutes—such as in the case of subsistence fisheries, welfare changes may be very sensitive to changes in ocean habitat.

9. Recommendations

The most important recommendation is the need to compile data useful for developing statistical models to estimate the value of ocean habitat. In the Saudi Arabian sector of the Red Sea, data currently are compiled on fish catches, fishing effort, and seafood prices. Additional data are needed to describe variations in ocean habitat across fishing regions and over time.

Measures of ocean habitat include mangrove acreage or coastal coverage, coral reef acreage, water quality, sea grass acreage, among other possibilities. Research on the ecological linkages between measures of habitat and the productivity of fish stocks can help inform the choice of measures for supply-side valuation. This research should include developing a better understanding of the extent to which the life cycles of exploited species depend upon the coral-seagrass-mangrove system. For many commercial species, research in tropical areas outside of the Red Sea environment could help inform the choice of habitat measures.

Once data are available, models must be developed to measure the marginal economic productivity or the resource rents of the Red Sea habitat. These models should be refined and updated on a regular basis as new data become available.

Results of the valuation should be made available for use in decisions about the economic consequences of human activities that may adversely impact the Red Sea habitat. For example, decisions to undertake coastal construction leading to the cutting of mangrove wetlands or the destruction of coral reefs should not be made without consideration of the opportunity costs, in terms of lost habitat productivity.

Decisions about coastal development should be informed by economic analysis, but they must also take into consideration the issues of fairness to existing users and the likely value of the resource to future generations. In cases where the net social benefits of coastal developments are positive, even if they lead to habitat degradation, further consideration should be made of the potential impacts on passive uses of the Red Sea environment. This consideration may require the application of a different set of methodological tools.

Acknowledgements

References

Aburto-Oropeza, O., Ezcurra, E., Danemann, G., Valdez, V., Murray, J., Sala, E., 2008. Mangroves in the Gulf of California increase fishery yields. PNAS 105, 10456–10459.

Allen, B.P., Loomis, J.B., 2006. Deriving values for the ecological support function of wildlife: an indirect valuation approach. Ecol. Econ. 56, 49–57.

Anderson, L.G., 1986. The Economics of Fisheries Management, second ed. Johns Hopkins University Press, Baltimore.

Armstrong, C.W., Falk-Peterson, J., 2008. Food for thought—habitat-fisheries interactions—a missing link? ICES J. Mar. Sci. 65, 817–821.

Barbier, E.B., 1994. Valuing environmental functions: tropical wetlands. Land Econ. 70, 155–173.

Barbier, E.B., 2003. Habitat-fishery linkages and mangrove loss in Thailand. Contemp. Econ. Pol'y 21, 59–77.

Barbier, E.B., 2007. Valuing ecosystem services as productive inputs. Econ. Pol'y. 22, 77–229.

Barbier, E.B., Strand, I., 1998. Valuing mangrove-fishery linkages: a case study of Campeche, Mexico. Environ. Res. Econ. 12, 151–166.

Barbier, E.B., Koch, E.W., Silliman, B.R., Hacker, S.D., Wolanski, E., Primavera, J., Granek, E.F., Polasky, S., Aswani, S., Cramer, L.A., Stoms, D.M., Kennedy, C.J., Bael, D., Kappel, C.V., Perillo, G.M.E., Reed, D.J., 2008. Coastal ecosystem-based management with nonlinear ecological functions and values. Science 319, 321–323.

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecol. Monogr. 81, 169–193.

Batie, S.S., Shabman, L.A., 1982. Estimating the economic value of wetlands: principles, methods, and limitations. Coast. Zone Manage. J. 10, 255–278.

Batie, S.S., Wilson, J.R., 1978. Economic values attributable to Virginia's coastal wetlands as inputs in oyster production. So. J. Ag. Econ. 10, 111–118.

Bell, F.W., 1989. Application of Wetland Valuation Theory to Florida Fisheries. SGR-95. Department of Economics, Florida State University, Tallahassee.

Bell, F.W., 1997. The economic valuation of saltwater marsh supporting marine recreational fishing in the southeastern United States. Ecol. Econ. 21, 243–254.

Berg, H., Öhman, M.C., Troëng, S., Lindén, O., 1998. Environmental economics of coral reef destruction in Sri Lanka. Ambio 27, 627–634.

Clark, C.W., 1990. Mathematical Bioeconomics. John Wiley and Sons Inc., New York, p. 400.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Rasking, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.

deGroot, R., Brander, L., van der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M., Crossman, N., Ghermandi, A., Hein, L., Hussain, S., Kumar, P., McVittie, A., Portela, R., Rodriguez, L.C., ten Brink, P., van Beukering, P., 2012. Global estimates of the value of ecosystems and their services in monetary units. Ecosys. Serv. 1, 50–61.

El Sayyed, M.K., 2008. Marine environment. In: Tolba, M.K., Saab, M.W. (Eds.), Arab Environment Future Challenges. Arab Federation for Environment and Development, Beirut, pp. 75–94, Ch. 6.

Ellis, G.M., Fisher, A.C., 1987. Valuing the environment as input. J. Environ. Manag. 25, 149–156.

Flaaten, O., 2011. Fisheries Economics and Management. Revision. Norwegian College of Fishery Science, Tromsø, p. 172 at: <https://www.ub.uit.no/munin/bitstream/handle/10037/2509/book.pdf?sequence=3> (last accessed 19 01 13.).

Freeman, A.M., 1991. Valuing environmental resources under alternative management regimes. Ecol. Econ. 3, 247–256.

Gladstone, W., Facey, R., Hariri, K., 2006. State of the Marine Environment: Report for the Red Sea and Gulf of Aden. Regional Organization for the Conservation of the Environment of the Red Sea and Gulf of Aden (PERSGA), Jeddah.

Gladstone, W, Curley, B., Shokri, M.R. 2012. Environmental impacts of tourism in the Gulf and the Red Sea. Mar. Poll Bull: http://dx.doi.org/10.1016/j.marpolbul.2012.09.017.

Gordon, H.S., 1954. The economic theory of a common-property resource: the fishery. J. Pol. Econ. 62, 124–142.

Gosselink, J.G., Odum, E.P., Pope, R.M., 1974. The Value of the Tidal Marsh. LSU-SG-74-03. Center for Wetland Resources, Louisiana State University, Baton Rouge.

Huettner, D.A., 1976. Net energy analysis: an economic assessment. Science 192, 101–104.

Jin, D., Kite-Powell, H.L., Hoagland, P., Solow, A.R., 2012. A bioeconomic analysis of traditional fisheries in the Red Sea. Mar. Res. Econ. 27, 137–148.

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.

Kellner, J.B., Sanchirico, J.N., Hastings, A., Mumby, P.J., 2011. Optimizing for multiple species and multiple values: tradeoffs inherent in ecosystem-based fisheries management. Conserv. Lett. 4, 21–30.

Kotb, M.M.A., 2010. The status of coral reefs in the Red Sea and Gulf of Aden: 2009. PERSGA Tech. Ser. No. 16. Jeddah: Regional Organization for the Conservation of the Environment of the Red Sea and Gulf of Aden (PERSGA), p. 125.

Lynne, G.D., Conroy, P., Prochaska, F.J., 1981. Economic valuation of marsh areas for marine production processes. J. Environ. Econ. Manag. 8, 175–186.

McArthur, L.C., Boland, J.W., 2006. The economic contribution of seagrass to secondary production in South Australia. Ecol. Model. 196, 163–172.

McConnell, K.E., Bockstael, N.E., 2005. Valuing the environment as a factor of production. In: Mäler, K.G., Vincent, J.R. (Eds.), . In: Handbook of Environmental Economics, vol. 2. Elsevier Science Publishing Company, Inc., New York, pp. 621–669.

McConnell, K.E., Strand, I.E., 1989. Benefits from commercial fisheries when demand and supply depend upon water quality. J. Environ. Econ. Manag. 17, 284–292.

Milon, J.W., 1989. Economic evaluation of artificial habitat for fisheries: progress and challenges. Bull. Mar. Sci. 44, 831–843.

Munro, G.R., Scott, A.D., 1985. The economics of fisheries management. In: Kneese, A.V., Sweeney, J.L. (Eds.), Handbook of Natural Resource and Energy Economics, Ch. 14. Elsevier Science Publishing Company, Inc., New York, pp. 623–676.

Regional Organization for the Conservation of the Environment of the Red Sea and Gulf of Aden (PERSGA), 2002. Status of the Living Marine Resources in the Red Sea and Gulf of Aden and Their Management. The World Bank, Washington.

Pope, R.M., Gosselink, J.G., 1973. A tool for use in making land management decisions involving tidal marshland. Coast. Manag. 1, 65–74.

Ragozin, D.L., Brown Jr., G., 1985. Harvest policies and nonmarket valuation in a predator-prey system. J. Environ. Econ. Manag. 12, 155–168.

Samonte-Tan, G.P.B., White, A.T., Tercero, M.A., Diviva, J., Tabara, E., Caballes, C., 2007. Economic valuation of coastal and marine resources: Bohol Marine Triangle, Philippines. Coast. Manag. 35, 319–338.

Sanchirico, J.N., Mumby, P., 2009. Mapping ecosystem functions to the valuation of ecosystem services: implications of species–habitat associations for coastal land-use decisions. Theor. Ecol. 2, 67–77.

Scott, A., 1955. The fishery: the objectives of sole ownership. J. Pol. Econ. 63, 116–124.

Shabman, L.A., Batie, S.S., 1978. Economic value of natural coastal wetlands: a critique. Coast. Zone Manage. J. 4, 231–247.

Swallow, S.K., 1990. Depletion of the environmental basis for renewable resources: the economics of interdependent renewable and nonrenewable resources. J. Environ. Econ. Manage 19, 281–296.

Tihansky, D.P., Meade, N.F., 1976. Economic contribution of commercial fisheries in valuing US estuaries. Coast. Zone Manage. J. 2, 411–421.

White, A.T., Vogt, H.P., Arin, T., 2000. Philippine coral reefs under threat: the economic losses caused by reef destruction. Mar. Poll. Bull. 40, 598–605.