Valuation of ecosystem services supporting aquatic and other land-based food systems

Edward B. Barbier[1]
Department of Economics and Finance, University of Wyoming, United States of America

Barbier, E.B. 2007. Valuation of ecosystem services supporting aquatic and other land-based food systems. In D.M. Bartley, C. Brugère, D. Soto, P. Gerber and B. Harvey (eds). *Comparative assessment of the environmental costs of aquaculture and other food production sectors: methods for meaningful comparisons.* FAO/WFT Expert Workshop. 24-28 April 2006, Vancouver, Canada. *FAO Fisheries Proceedings*. No. 10. Rome, FAO. 2007. pp. 71–86

ABSTRACT

Increased aquaculture production globally will require both land and water, placing additional stress on natural ecosystems. A major concern of the growth in intensive livestock production is that the resulting animal waste is overburdening the assimilative capacity of aquatic ecosystems, disrupting their provision of valuable services. This paper explains the economic approach to valuing ecosystem services generally, and especially those ecosystem regulatory and habitat functions that support aquatic and other land-based food systems. The paper uses the specific example of ecological support services for aquaculture in Thailand as a case study to illustrate some key approaches.

INTRODUCTION

Over the past three decades, global output from aquaculture grew at an annual average rate of 9.1 percent, reaching 39.8 million metric tons in 2002 (FAO, 2005). This growth rate was higher than any other animal food-producing systems, including livestock rearing for meat. By 2020, the baseline projection for global aquaculture production is 53.6 million metric tons, but could be as high as 83.6 million metric tons (Delgado *et al.*, 2003).

In recent decades, global livestock production, particularly of cattle, swine and poultry, has undergone a major change towards industrialization. The most important trend has been the relocation of livestock from pastures, lots and pens into large buildings where the animals are confined and fed until they are ready for market. Such confined livestock units have spurred the global increase in production through intensive feedstuffs and reducing land constraints (Gollehon *et al.*, 2001; Mallin and Cahoon, 2003). As a result, industrial livestock farming systems are growing at twice the rate of traditional mixed farming systems and six times as fast as grazing-based systems (FAO, 2000).

[1] ebarbier@uwyo.edu

These trends in aquatic and intensive livestock production have important environmental implications, especially for the ecosystem services supporting the production.

Increased aquaculture production globally will require both land and water, placing additional stress on natural ecosystems. There is also concern about the environmental impacts of intensive systems, especially the large-scale production required for shrimp, salmon and other high-valued species. For instance, aquaculture accounts for 52 percent of mangrove loss globally, with shrimp farming alone accounting for 38 percent of mangrove deforestation (Valiela, Bowen and York, 2001). Accompanying the loss of these coastal habitats is the loss of a range of vital ecosystem services, ranging from nurseries for fish fry to storm protection. Intensive aquaculture systems can also lead to water shortages and pollution from effluent discharges, disrupting the functioning of coastal and aquatic ecosystems through nutrient overload (Goldburg and Naylor, 2005). A major concern of the growth in intensive livestock production is that the resulting animal waste is overburdening the assimilative capacity of aquatic ecosystems (Gollehon *et al.*, 2001; Mallin and Cahoon, 2003), disrupting their provision of valuable services. Excessive animal manure causes a range of environmental problems for these systems, including nitrate, phosphate and ammonia pollution, increased biological oxygen demand (BOD), algal blooms and eutrophication, and contamination by fecal pathogens. The resulting loss of ecological services ranges from the destruction of aquatic fish habitats and nursery grounds, to loss of potable water supplies to human health impacts to loss of recreational and aesthetic benefits, to effects on property values.

Another important ecological support service for much farmed fish is its dependence on marine fish, such as anchovies, sardines, capelin and other lower trophic species, for the fish meal and oils used in feeds. Increased growth in aquaculture may mean increasing pressure on the "export support service" of marine fisheries supplying the input species used in feeds (Delgado *et al.*, 2003; Naylor *et al.*, 2000). Finally, there is growing concern that marine fish farming may increase the risk of invasive species problems in surrounding ecosystems through the increased number of escaped farm fish that interact with wild fish (Goldburg and Naylor, 2005).

The purpose of the following paper is to explain why valuing these ecological support services for aquaculture and intensive livestock production will become increasingly important as these systems expand globally. The first part of the paper will explore the economic approach to valuing ecosystem services generally. The second part of the paper will discuss the various methods available to value ecosystem services and uses the specific example of shrimp aquaculture in Thailand as an illustration. The paper concludes by examining further research issues in the valuation of ecological services that support aquatic and other food production systems.

What are ecosystem services?

Broadly defined, "ecosystem services are the benefits people obtain from ecosystems" (Millennium Ecosystem Assessment 2003, p. 53). Such benefits are typically described by ecologists in the following manner: "Ecosystem services are the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life. In addition to the production of goods, ecosystem system services are the actual life-support functions, such as cleansing, recycling, and renewal, and they confer many intangible aesthetic and cultural benefits as well." (Daily, 1997, p. 3). Thus in the current literature the term "ecosystem services" lumps together a variety of "benefits", which in economics would normally be classified under three different categories: (i) "goods" (e.g. products obtained from ecosystems, such as resource harvests, water and genetic material), (ii) "services" (e.g. recreational and tourism benefits or certain ecological regulatory functions, such as water purification,

TABLE 1
Some services provided by ecosystem regulatory and habitat functions

Ecosystem functions	Ecosystem processes and components	Ecosystem services (benefits)
Regulatory Functions		
Gas regulation	Role of ecosystems in biogeochemical processes	Ultraviolet-B protection Maintenance of air quality Influence of climate
Climate regulation	Influence of land cover and biologically mediated processes	Maintenance of temperature, precipitation
Disturbance prevention	Influence of system structure on dampening environmental disturbance	Storm protection Flood mitigation
Water regulation	Role of land cover in regulating runoff, river discharge and infiltration	Drainage and natural irrigation Flood mitigation Groundwater recharge
Soil retention	Role of vegetation root matrix and soil biota in soil structure	Maintenance of arable land Prevention of damage from erosion and siltation
Soil formation	Weathering of rock and organic matter accumulation	Maintenance of productivity on arable land
Nutrient regulation	Role of biota in storage and recycling of nutrients	Maintenance of productive ecosystems
Waste treatment	Removal or breakdown of nutrients and compounds	Pollution control and detoxification
Habitat Functions		
Niche and refuge	Suitable living space for wild plants and animals	Maintenance of biodiversity Maintenance of beneficial species
Nursery and breeding	Suitable reproductive habitat and nursery grounds	Maintenance of biodiversity Maintenance of beneficial species

Sources: Adapted from Heal *et al.* (2005, Table 3-3) and De Groot, Wilson and Boumans (2002).

climate regulation, erosion control, etc.), and (iii) cultural benefits (e.g., spiritual and religious, heritage, etc.).

Regardless how one defines and classifies "ecosystem services", as a report from The US National Academy of Science has emphasized, "the fundamental challenge of valuing ecosystem services lies in providing an explicit description and adequate assessment of the links between the structure and functions of natural systems, the benefits (i.e., goods and services) derived by humanity, and their subsequent values" (Heal *et al.*, 2005, p. 2). Moreover, it has been increasingly recognized by economists and ecologists that the greatest "challenge" they face is in valuing the ecosystem services provided by a certain class of key ecosystem functions – regulatory and habitat functions. Table 1 provides some examples of the links between regulatory and habitat functions and the ecosystem services that ultimately benefit humankind.

The valuation challenge

The literature on ecological services implies that natural ecosystems are assets that produce a flow of beneficial goods and services over time. In this regard, they are no different from any other asset in an economy, and in principle, ecosystem services should be valued in a similar manner. That is, regardless of whether or not there exists a market for the goods and services produced by ecosystems, their social value must equal the discounted net present value (NPV) of these flows.

For example, letting B_t be the social benefits in any time period t, from ecosystem services, then the social value of these flows is:

$$V_0 = \sum_0^T \frac{B_t}{(1+r)^t} \tag{1}$$

where r is the social rate of discount. In addition, just as for any economic asset, B_t, can be measured by the aggregate willingness to pay by the individuals benefiting in each period from ecosystem services.

However, what makes environmental assets special is that they give rise to particular measurement problems that are different than those for conventional economic or financial assets. This is especially the case for the beneficial services that are derived from the regulatory and habitat functions of natural ecosystems.

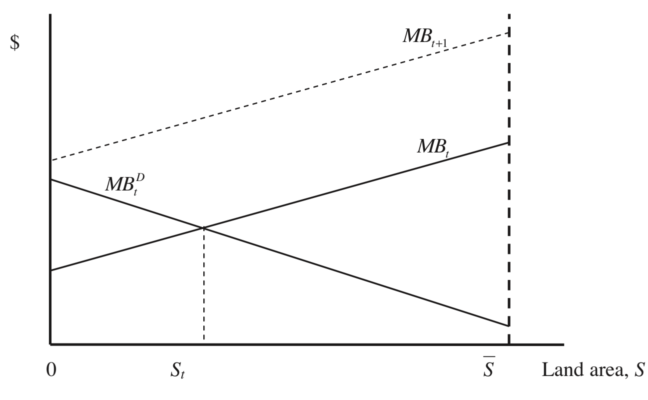
For one, these assets and services fall in the special category of "nonrenewable resources with renewable service flows" (Just, Hueth and Schmitz, 2004, p. 603). Although a natural ecosystem providing such beneficial services is unlikely to increase, it can be depleted, e.g. through habitat destruction, land conversion, pollution impacts and so forth. Nevertheless, if the ecosystem is left intact, then the flow services from the ecosystem's regulatory and habitat functions are available in quantities that are not affected by the rate at which they are used.

In addition, whereas the services from most assets in an economy are marketed, the benefits arising from the regulatory and habitat functions of natural ecosystems generally are not. If the aggregate willingness to pay for these benefits, B_t, is not revealed through market outcomes, then efficient management of such ecosystem services requires explicit methods to measure its social value (e.g., see Freeman, 2003; Just, Hueth and Schmitz, 2004).

A further concern over ecosystem services is that their beneficial flows are threatened by the widespread disappearance of natural ecosystems and habitats across the globe (Millennium Ecosystem Assessment, 2003). As noted in the introduction, aquatic and other land-based food systems are an important cause of this disappearance, due to increased demand for land and pollution. The failure to measure explicitly the aggregate willingness to pay for otherwise non-marketed ecological services exacerbates these problems, as the benefits of these services are "underpriced" and may lead to excessive land conversion, habitat fragmentation and pollution caused by aquatic and other land-based food systems.

Figure 1 illustrates the difficulty that these environmental measurement problems pose. Assume that at any time t, the marginal social benefits of ecological services are represented by the line MB_t for a natural ecosystem area of given area $\overline{S}$. The aggregate willingness to pay for the benefits of these services, B_t, is simply the area under this curve. If there is no other use for the land occupied by the ecosystem, then the opportunity costs of maintaining it are zero, and the ecosystem will be left intact and continue to provide the same flow of services in perpetuity. However, population and economic development pressures in many areas of the world usually mean that the opportunity cost of maintaining the land for natural ecosystems is not zero, due to increased demand for land for aquatic and other land-based food systems. Suppose that the marginal social benefits of converting natural ecosystem land for these development options is represented by MB_t^D in the figure. Thus efficient use of land would require that an amount S_t of ecosystem area should be converted for food systems leaving $\overline{S} - S_t$ of the natural ecosystem intact.

FIGURE 1
Optimal ecosystem conversion to aquatic and land-based food systems



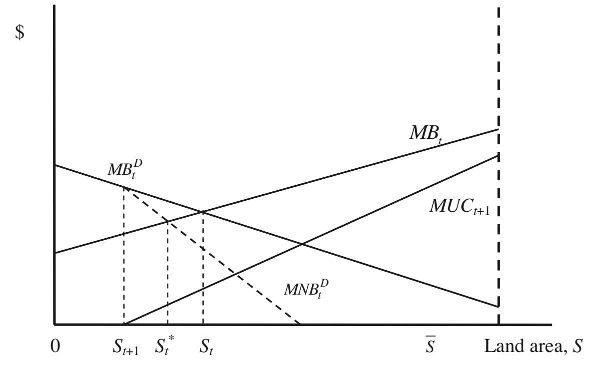
Both of these outcomes assume that the willingness to pay for the marginal benefits arising from ecosystem services, MB_t, is explicitly measured, or "valued". But if this is not the case, then these non-marketed flows are likely to be ignored in the land use decision. Only the marginal benefits, MB_t^D, of the "marketed" outputs arising from aquatic and other land-based food systems will be taken into account, and as indicated in the figure, this implies that the entire ecosystem area $\overline{S}$ will be converted for development.

A further problem in valuing environmental assets is the uncertainty over their future values. It is possible, for example, that the benefits of natural ecosystem services are larger in the future as more scientific information becomes available over time. This is illustrated in Figure 1. As has been already shown, based on the valuation of marginal benefits of ecosystem services in the current period, amount S_t of ecosystem area should be optimally converted for aquatic and land-based food systems at time t. However, suppose that in some future period t+1 it is discovered that the value of ecosystem services is actually much larger, so that the discounted marginal benefits of these services, MB_{t+1}, is now represented by the dotted line in the figure. If the discounted marginal benefits from aquatic and other food systems in the future are largely unchanged, i.e. $MB_{t+1}^D \approx MB_t^D$, then as Figure 1 indicates, the discounted future benefits of ecosystem services exceed these costs, and the ecosystem area should be "restored" to its original area $\overline{S}$.

The need to consider future ecosystem service values is further exacerbated by the problem of irreversibility. As pointed out by Krutilla and Fisher (1985), if environmental assets are irreversibly depleted, their value will rise relative to the value of other reproducible and accumulating economic assets. This is particularly the case for any natural ecosystem that is irreversibly converted or degraded as a result of expansion of aquatic and other land-based food systems or the cumulative generation of pollution by these systems. Because natural ecosystems are in fixed supply and are difficult to substitute for or restore, the beneficial services provided by their regulatory and habitat functions will decline as these assets are converted or degraded. The increasing relative scarcity of these services means that their value will rise relative to other goods and services in the economy. This also implies that any decision today that leads to irreversible conversion imposes a "user cost" on individuals who face a rising scarcity value of future ecosystem benefits as a consequence.

Figure 2 illustrates the additional measurement problem arising from irreversible conversion of fixed ecosystem assets. As in the original example of Figure 1, if only the current benefits, MB_t, and opportunity costs, MB_t^D , of maintaining a natural ecosystem are considered, then an amount S_t of ecosystem area would be converted today. But suppose that the loss of ecosystem services arising from converting S_t causes the value of these services to rise. As a result, individuals benefiting from these services in a future time period t+1 would choose optimally to have less land converted to aquatic and other food systems, i.e. $S_{t+1} < S_t$. However, if ecosystem conversion is irreversible, then land development for food systems remains at S_t in time period t+1. The welfare effect of the reduced choice for individuals in the future is the user cost of irreversible loss of ecosystem services, which in present value terms is represented as MUC_{t+1} in the figure. The correct land use decision would take into account this additional cost of irreversible ecosystem conversion due to expansion of aquatic and other food systems today. Deducting the marginal user cost from MB_t^D yields the net marginal benefits of the development option, MNB_t^D . The latter is the appropriate measure of the opportunity

FIGURE 2
Irreversible conversion of ecosystems and uncertainty



costs of maintaining the ecosystem, and equating it with the marginal social benefits of ecosystem services determines the intertemporally optimal land allocation. Only S_t^* of ecosystem area should be converted for aquatic and other land-based food systems leaving $\overline{S} - S_t^*$ of the natural ecosystem intact.

Valuation of environmental assets under conditions of uncertainty and irreversibility clearly poses additional measurement problems. There is now a considerable literature advocating various methods for estimating environmental values by measuring the additional amount, or "premium" that individuals are willing to pay to avoid the uncertainty surrounding such values (see Ready 1995 for a review). Similar methods are also advocated for estimating the user costs associated with irreversible development, as this also amounts to valuing the "option" of avoiding reduced future choices for individuals (Just, Hueth and Schmitz, 2004). However, the problem with such welfare measures is that they cannot be estimated from the observed behaviour of individuals and are therefore difficult to implement empirically, particularly when there is uncertainty not only about the future state of the environmental asset but also over the future preferences and income of individuals. The general conclusion from the few empirical attempts to implement environmental valuation under uncertainty is that "more empirical research is needed to determine under what conditions we can ignore uncertainty in benefit estimation….where uncertainty is over economic parameters such as prices or preferences, the issues surrounding uncertainty may be empirically unimportant" (Ready, 1995, p. 590).

Valuation methods

Uncertainty and irreversible loss are important issues to consider in valuing ecosystem services affected by aquatic and other land-based food systems. However, as emphasized by Heal *et al.,* (2005), a "fundamental challenge" in valuing these flows is that ecosystem services are largely not marketed, and unless some attempt is made to value the aggregate willingness to pay for these services, B_t, then management of natural ecosystems and their services will not be efficient.

In recent years substantial progress has been made by economists working with ecologists and other natural scientists on this "fundamental challenge" to improve environmental valuation methodologies. Table 2 indicates there are now various methods that can be used for valuing the services derived from ecological regulatory and habitat functions. It is beyond the scope of this paper, however, to discuss all the valuation methods listed in Table 2. More discussion of the methods and their application to valuing ecosystem goods and services can be found in Freeman (2003), Heal *et al.,* (2005) and Pagiola, von Ritter and Bishop (2004). Instead, this section will make a few observations concerning these valuation methods, emphasizing both their advantages and shortcomings.

First, the application of some valuation methods is often limited to specific types of ecological services. For example, the travel cost method is used principally for those environmental values that enhance individuals' enjoyment of recreation and tourism, averting behaviour models are best applied to the health effects arising from environmental pollution and hedonic wage and property models are used primarily for assessing work-related environmental hazards and environmental impacts on property values, respectively.

In contrast, stated preference methods, which include contingent valuation methods, conjoint analysis and choice experiments, have the potential to be used widely in valuing ecosystem goods and services. These valuation methods share the common approach of surveying individuals who benefit from an ecological service or range of services, in the hope that analysis of these responses will provide an accurate measure of the individuals' willingness to pay for the service or services. In addition, stated preference methods can go beyond estimating the value to individuals of single and

TABLE 2
Various valuation methods applied to ecosystem services

Valuation method[a]	Types of value estimated[b]	Common types of applications	Ecosystem services valued
Travel cost	Direct use	Recreation	Maintenance of beneficial species, productive ecosystems and biodiversity
Averting behaviour	Direct use	Environmental impacts on human health	Pollution control and detoxification
Hedonic price	Direct and indirect use	Environmental impacts on residential property and human morbidity and mortality	Storm protection; flood mitigation; maintenance of air quality
Production function	Indirect use	Commercial and recreational fishing; agricultural systems; control of invasive species; watershed protection; damage costs avoided	Maintenance of beneficial species; maintenance of arable land and agricultural productivity; prevention of damage from erosion and siltation; groundwater recharge; drainage and natural irrigation; storm protection; flood mitigation
Replacement cost	Indirect use	Damage costs avoided; freshwater supply	Drainage and natural irrigation; storm protection; flood mitigation
Stated preference	Use and non-use	Recreation; environmental impacts on human health and residential property; damage costs avoided; existence and bequest values of preserving ecosystems	All of the above

Notes:
[a] See Freeman (2003), Heal *et al.* (2005) and Pagiola, von Ritter and Bishop (2004) for more discussion of these various valuation methods and their application to valuing ecosystem goods and services.
[b] Typically, use values involve some human "interaction" with the environment whereas non-use values do not, as they represent an individual valuing the pure "existence" of a natural habitat or ecosystem or wanting to "bequest" it to future generations. Direct use values refer to both consumptive and non-consumptive uses that involve some form of direct physical interaction with environmental goods and services, such as recreational activities, resource harvesting, drinking clean water, breathing unpolluted air and so forth. Indirect use values refer to those ecosystem services whose values can only be measured indirectly, since they are derived from supporting and protecting activities that have directly measurable values.
Source: Adapted from Heal *et al.* (2005), Table 4-2.

even multiple benefits of ecosystems and in some cases elicit "non-use values", i.e. the additional "existence" and "bequest" values that individuals attach to ensuring that a preserved and well-functioning system will be around for future generations to enjoy. For example, a study of mangrove-dependent coastal communities in Micronesia demonstrated through the use of contingent valuation techniques that the communities "place some value on the existence and ecosystem functions of mangroves over and above the value of mangroves' marketable products" (Naylor and Drew, 1998, p. 488). Similarly, choice experiments and conjoint analysis, which ask respondents to rank, rate or choose among various environmental outcomes or scenarios, have the potential to elicit the relative values that individuals place on different ecosystem services (see for example Carlsson, Frykblom and Lijenstolpe, 2003).

However, as emphasized by Heal *et al.* (2005), to implement a stated-preference study two key conditions are necessary: (1) the information must be available to describe the change in a natural ecosystem in terms of service that people care about, in order to place a value on those services; and (2) the change in the natural ecosystem must be explained in the survey instrument in a manner that people will understand and not reject the valuation scenario. For many of the specific services arising from the type of ecological regulatory and habitat functions listed in Table 1, one or both of these conditions may not hold. For instance, it has proven very difficult to describe accurately through the hypothetical scenarios required by stated-preference surveys how changes in ecosystem processes and components affect ecosystem regulatory and habitat functions and thus the specific benefits arising from these functions that individuals value. If there is considerable scientific uncertainty surrounding these linkages, then not only is it difficult to construct such hypothetical scenarios but also any responses elicited from individuals from stated-preference surveys are likely to yield inaccurate measures of their willingness to pay for ecological services.

In contrast to stated-preference methods, the advantage of production function (PF) approaches is that they depend on only the first condition, and not both conditions, holding. That is, for those regulatory and habitat functions where there is sufficient scientific knowledge of how these functions link to specific ecological services that support or protect economic activities, then it may be possible to employ the PF approach to value these services. The basic modelling approach underlying PF methods, also called "valuing the environment as input", is similar to determining the additional value of a change in the supply of any factor input (Barbier, 1994 and 2000; Freeman, 2003). If changes in the regulatory and habitat functions of ecosystems affect the marketed production activities of an economy, then the effects of these changes will be transmitted to individuals through the price system via changes in the costs and prices of final good and services. This means that any resulting "improvements in the resource base or environmental quality" as a result of enhanced ecosystem services, "lower costs and prices and increase the quantities of marketed goods, leading to increases in consumers' and perhaps producers' surpluses" (Freeman, 2003, p. 259).

An adaptation of the PF methodology is required in the case where ecological regulatory and habitat functions have a protective value, through various ecological services such as storm protection, flood mitigation, prevention of erosion and siltation, pollution control and maintenance of beneficial species (Table 1). In such cases, the environment may be thought of producing a non-marketed service, such as "protection" of economic activity, property and even human lives, which benefits individuals through limiting damages. Applying PF approaches requires modelling the "production" of this protection service and estimating its value as an environmental input in terms of the expected damages avoided by individuals (Barbier, 2006).

However, PF methods have their own measurement issues and limitations. For instance, applying the PF method raises questions about how changes in the ecological service should be measured, whether market distortions in the final goods market are significant, and whether current changes in ecological services may affect future productivity through biological "stock effects". A common approach in the literature is to assume that an estimate of ecosystem area may be included in the "production function" of marketed output as a proxy for the ecological service input. For example, this is the standard approach adopted in coastal habitat-fishery PF models, as allowing wetland area to be a determinant of fish catch is thought by economists and ecologists to proxy some element of the productivity contribution of this important habitat function (Barbier, 2000; Freeman, 2003). In addition, as pointed out by Freeman (1991), market conditions and regulatory policies for the marketed output will influence the values imputed to the environmental input. For instance, in the previous example of coastal wetlands supporting an offshore fishery, the fishery may be subject to open access conditions. Under these conditions, profits in the fishery would be dissipated, and price would be equated to average and not marginal costs. As a consequence, producer values are zero and only consumer values determine the value of increased wetland area. Finally, a further measurement issue arises in the case where the ecological service supports a natural resource system, such as a fishery, forestry or a wildlife population, which is then harvested or exploited through economic activity. In such cases, the key issue is whether or not the effects on the natural resource stock or biological population of changes in the ecological service are sufficiently large that these stock effects need to be modelled explicitly. In the production function valuation literature, approaches that ignore stock effects are referred to as "static models" of environmental change on a natural resource production system, whereas approaches that take into account the intertemporal stock effects of the environmental change are referred to as "dynamic models" (Barbier, 2000; Freeman, 2003).

In circumstances where an ecological service is unique to a specific ecosystem and is difficult to value, then economists have sometimes resorted to using the cost of replacing the service or treating the damages arising from the loss of the service as a valuation approach. Economists consider that the replacement cost approach should be used with caution. For example, the few studies that have attempted to value the storm prevention and flood mitigation services of the "natural" storm barrier function of mangrove systems have employed the replacement cost method by simply estimating the costs of replacing mangroves by constructing physical barriers to perform the same services (Chong, 2005). Shabman and Batie (1978) suggested that this method can provide a reliable valuation estimation for an ecological service, but only if the following conditions are met: (1) the alternative considered provides the same services; (2) the alternative compared for cost comparison should be the least-cost alternative; and (3) there should be substantial evidence that the service would be demanded by society if it were provided by that least-cost alternative. Unfortunately, very few replacement cost studies meet all three conditions.

In the absence of conducting reliable stated preference surveys to elicit the willingness to pay by individuals for ecological services, for some benefits an alternative to employing either replacement cost or cost of treatment methods might be the expected damage function (EDF) approach. The EDF approach is nominally straightforward; it assumes that the value of an asset that yields a benefit in terms of reducing the probability and severity of some economic damage is measured by the reduction in the expected damage. The essential step to implementing this approach, which is to estimate how changes in the asset affect the probability of the damaging event occurring, has been used routinely in risk analysis and health economics, e.g. as in the case of airline safety performance, highway fatalities, drug safety and studies of the incidence of diseases and accident rates (Cameron and Trivedi, 1998; Winkelmann, 2003). Barbier (2006) shows that the EDF approach can be applied, under certain circumstances, to value ecological services that also reduce the probability and severity of economic damages, such as the storm protection service of mangroves.

Valuation of ecosystem services supporting aquaculture in Thailand

Since 1961, Thailand has lost from 1 500 to 2 000 km^2 of coastal mangroves, or about 50–60 percent of the original area (Wilkie and Fortuna, 2003). Over 1975–96, 50–65 percent of Thailand's mangroves were lost to shrimp farm conversion alone (Aksornkoae and Tokrisna, 2004).

Mangrove deforestation in Thailand has focused attention on the two principle services provided by mangrove ecosystems, their role as nursery and breeding habitats for off-shore fisheries and as natural "storm barriers" to periodic coastal storm events, such as wind storms, tsunamis, storm surges and typhoons. In addition, many coastal communities exploit mangroves directly for a variety of products, such as fuelwood, timber, raw materials, honey and resins, and crabs and shellfish. One study estimated that the annual value to local villagers of collecting these products was US$88 per hectare (ha), or approximately US$823/ha in net present value terms over a 20-year period and with a 10 percent discount rate (Sathirathai and Barbier, 2001). The same study also used the "replacement cost" method of estimating the value of the protection service of mangrove ecosystems and a "static" habitat-fishery model to estimate their role in supporting offshore fisheries. To compare these benefits, the authors also estimated the economic returns to shrimp farming that converts mangrove area, which included an estimate of the water pollution damages and the costs of replanting lost mangroves.

The above economic costs of maintaining shrimp aquaculture in Thailand suggest that the net benefits of this activity need to be compared to the economic benefits of the ecosystem services of the mangrove area that is converted to shrimp farming. Only by comparing the returns to these two alternative uses is it possible to determine whether or not full conversion of mangroves into commercial shrimp farms is worthwhile (Figure 1).

Several analyses have demonstrated that the overall commercial profitability of shrimp aquaculture in Thailand provides a substantial incentive for private landowners to invest in such operations (Barbier, 2003; Sathirathai and Barbier, 2001; Tokrisna, 1999). However, many of the conventional inputs used in shrimp pond operations are subsidized, below border-equivalent prices, thus increasing artificially the private returns to shrimp farming. Thus the first step in the analysis of the net benefits of shrimp aquaculture is to adjust the costs of the activity for these subsidies. The results of this calculation are shown in Table 3.

The productive life of a typical commercial shrimp farm in Southern Thailand is normally five years. After this period, there tends to be problems of drastic yield

TABLE 3
Economic returns to shrimp aquaculture, Thailand (1996 US$)

	Year						
Value(US$)/ha	**1**	**2**	**3**	**4**	**5**	**6**	**7-20**
Benefits Gross Returns[a]	20 719	20 719	20 719	20 719	20 719		
Costs Variable costs[b] Annualized fixed costs[c]	 16 800 3 597	 16 800 3 597	 16 800 3 597	 16 800 3 597	 16 800 3 597		
Cost of pollution[d]	264	264	264	264	264		
Costs of forest rehabilitation[e]						9 521	137
Net economic returns:							
Net present value (10 percent discount rate)	1 341.48						
Net present value (12 percent discount rate)	1 298.85						
Net present value (15 percent discount rate)	1 240.18						
With pollution control:							
Net present value (10 percent discount rate)	241.90						
Net present value (12 percent discount rate)	234.21						
Net present value (15 percent discount rate)	223.63						
With forest rehabilitation:							
Net present value (10 percent discount rate)	–6 294.79						
Net present value (12 percent discount rate)	–5 682.04						
Net present value (15 percent discount rate)	–4 898.76						

Notes:

[a] Assumes non-declining yields over five-year period of investment, and based on estimates of average shrimp yields of 3,856.25 kg/ha and farm price (1996$) of $5.373/kg.

[b] Includes costs of shrimp larvae, feed, gasoline, oil and electricity, pond cleaning, pond and machine maintenance, labor and miscellaneous variable costs, which are adjusted using the standard conversion factor of 0.89 for operating costs in Thailand.

[c] Land tax and rent, interest payments, opportunity cost of land and pond clearing costs, and depreciation, which are adjusted using the standard conversion factor of 0.961 for capital costs in Thailand.

[d] Based on costs of treatment of chemical pollutants in water and loss of farm income from rice production from saline water released from shrimp ponds.

[e] Based on costs of rehabilitating abandoned shrimp farms, replanting mangrove forests and maintaining and protecting mangrove seedlings.

Source: Adapted from Sathirathai and Barbier (2001).

decline and disease; shrimp farmers then usually abandon their ponds and find a new location. The gross returns of aquaculture are, nonetheless, very high – around US$20 719 per hectare per year in real terms (Table 3). With the operating and capital costs adjusted for subsidies, the discounted economic returns range from US$1 240 to US$1 341 per hectare.

In addition, a major external cost of shrimp ponds is the considerable amount of water pollution that they generate. This consists of both the high salinity content of water released from the ponds and agrochemical runoff. When the costs of controlling pollution are taken into account, the annual net benefits of shrimp farms fall to $58 per hectare, and the discounted net returns from aquaculture decline to US$224 to US$242 per hectare (Table 3).

There is also the problem of the highly degraded state of abandoned shrimp ponds after the five-year period of their productive life. Across Thailand those areas with abandoned shrimp ponds degenerate rapidly into wasteland, since the soil becomes very acidic, compacted and too poor in quality to be used for any other productive use, such as agriculture. This reflects the fact that converting mangroves to establish shrimp farms is an "irreversible" land use, and without considerable additional investment in restoration, these areas do not regenerate into mangrove forests. Thus one approach to account for this "user cost" of converting mangroves irreversibly is to incorporate this cost explicitly in the estimation of the net returns to shrimp aquaculture. However, as shown in Table 3, these restoration costs are considerable, and mean that the shrimp aquaculture operation makes an economic loss.

An important issue is whether it is worth restoring mangroves in the first place. If the foregone benefits of the ecological services of mangroves are not large, then mangrove restoration may not be a reasonable option. Table 4 indicates the value of three of these benefits: the net income from local mangrove forest products, habitat-fishery linkages and storm protection.

Sathirathai and Barbier (2001) estimate the value to local communities of using mangrove resources in terms of the net income generated by various wood and non-wood products from forests. If the extracted products were sold, market prices were used to calculate the net income generated (gross income minus the cost of extraction). If the products were used only for subsistence, the gross income was estimated based on surrogate prices, i.e., the market prices of the closest substitute. Based on surveys of local villagers in Surat Thani Province, the major products collected by the households were various fishery products, honey, and wood for fishing gear and fuelwood. As shown in Table 4, the net annual income from these products is $101 per hectare. Although this is the lowest benefit generated by mangrove forests, this value is still nearly twice as much as the net annual economic returns from shrimp aquaculture once the costs of pollution control are taken into account.

TABLE 4

Net present value of mangrove forest benefits, Thailand (1996 US$)[a]

	Value(US$)/ha
Net income from timber and non-timber products[b]	101.49
Habitat-fishery linkages[c]	248.70
Storm protection[d]	1 878.98
Total benefits	**2 229.17**
Net present value (10 percent discount rate)	20 876.00
Net present value (12 percent discount rate)	18 648.74
Net present value (15 percent discount rate)	16 046.08

Notes:
[a] All benefits estimated on an annual basis; net present value calculations are based on a 20-year time horizon.
[b] Adapted from Sathiathai and Barbier (2001).
[c] From Barbier (2003), assuming a price elasticity of demand for fish of -0.5.
[d] From Barbier (2006).

Barbier (2003) shows how the coastal habitat-fishery of mangroves in Thailand may be modelled through incorporating the change in wetland area within a multi-period harvesting model of the fishery. The key to this approach is to model a coastal wetland that serves as a breeding and nursery habitat for fisheries as affecting the growth function of the fish stock. As a result, the value of a change in this habitat-support function is determined in terms of the impact of any change in mangrove area in the long run equilibrium conditions of the fishery. As Table 4 indicates, the net annual benefit of this service is $249 per hectare.

The methodology of the EDF valuation approach is described in Barbier (2006) for estimating the expected damage costs avoided through increased provision of the storm protection service of coastal wetlands. Two components are critical to implementing the EDF approach to estimating the changes in expected storm damages: the influence of wetland area on the expected incidence of economically damaging natural disaster events, and some measure of the additional economic damage incurred per event. Both of these components can be estimated, provided that there are sufficient data on past storm events, and preferably across different coastal areas, as well as estimates of the economic damages inflicted by each event. The most important step in the analysis is the first one, and provided that there is sufficient data on the incidence of past natural disasters and changes in wetland area in coastal regions, this step can be done through employing a count data model (Cameron and Trivedi, 1998; Winkelmann, 2003). The EDF approach is then applied to estimate the benefits from the storm protection service of mangroves in Thailand, which is calculated to be $1 879 per hectare (Table 4).

Table 4 indicates that the total annual sum of these three mangrove benefits is $2 229 per ha in constant 1996 prices. The value of the storm protection service clearly dominates these benefits. However, each one of these benefits has an annual value in excess of the annual economic returns from shrimp aquaculture net of pollution control costs. The net present value of all three mangrove ecosystem benefits ranges from $16 046 to $20 876 per hectare.

To summarize, this case study has shown the importance of valuing the ecological services that support aquaculture systems. Controlling the pollution generated by aquaculture generates substantial costs, and these must be taken into account in any economic assessment of the economic returns from aquaculture. In addition, the irreversible conversion of mangroves for aquaculture results in the loss of ecological services that generate significantly large economic benefits. This loss of benefits must be taken into account in land use decisions that lead to the widespread conversion of mangroves. Finally, the largest economic benefits of mangroves appear to arise from regulatory and habitat functions. This reinforces the importance of measuring the value of such ecological services.

CONCLUSIONS

Important advances have been made recently in the economic valuation of key ecological services supporting aquaculture and other land-based food systems. This paper has reviewed some of the key approaches.

The Thailand case study in this paper does not suggest that shrimp aquaculture should be halted in Thailand. It does suggest, however, the need for better policies to control excessive shrimp farm expansion and subsequent mangrove loss by making aquaculture in Thailand more sustainable. To achieve this objective, there are clearly several steps that the Government of Thailand could take to reduce the current perverse incentives for excessive mangrove conversion for shrimp farming. These include eliminating preferential subsidies for the inputs, such as larvae, chemicals and machinery, used in shrimp farming, ending preferential commercial loans for clearing land and establishing shrimp ponds, employing land auctions and concession fees for the establishment of new farms in the “economic zones” of coastal areas, and finally, charging replanting fees for farms that convert mangroves (Barbier and Sathirathai, 2004). Reducing the other environmental impacts of shrimp farming in Thailand is also important, notably problems of water pollution, the depletion of wild fish stocks for feed and disease outbreaks within ponds (Goldberg and Naylor, 2005; Jory, 1996; Naylor *et al.,* 2000). As one industry expert has commented: "the key to industry sustainability in Thailand, as it is for most shrimp farming countries, is continuing research and breakthrough in three areas: species domestication, minimizing the negative environmental impact of pond effluents on coastal ecosystems, and controlling diseases, especially those caused by viruses" (Jory 1996, p. 74).

Although this paper focused on the specific example of the ecological support services of shrimp aquaculture only, it is clear that the valuation of ecosystem services should be applied to the environmental impacts of other land-based food systems. For instance, the animal waste from the growth in intensive livestock production is overburdening the assimilative capacity of aquatic ecosystems (Gollehon *et al.,* 2001; Mallin and Cahoon, 2003). The result is disruption of ecological services ranging from the destruction of aquatic fish habitats and nursery grounds to loss of potable water supplies, to human health impacts, to loss of recreational and aesthetic benefits, to effects on property values. All these foregone ecological benefits can and should be valued to assess the damages arising from the expansion of the intensive livestock industry.

In sum, this paper has shown that valuing the non-market benefits of ecological regulatory and habitat services is becoming increasingly important in assisting policy makers in the management of critical environmental assets that support aquaculture and other land-based food systems. However, further progress applying production function approaches and other methods to value ecological services faces two challenges.

First, for these methods to be applied effectively to valuing ecosystem services, it is important that the key ecological and economic relationships are well understood. Unfortunately, our knowledge of the ecological functions, let alone the ecosystem processes and components, underlying many of the services listed in Table 1 is still incomplete.

Second, natural ecosystems are subject to stresses, rapid change and irreversible losses, they tend to display threshold effects and other non-linearities that are difficult to predict, let alone model in terms of their economic impacts. These uncertainties can affect the estimation of values from an *ex ante* ("beforehand") perspective, which is the perspective adopted by the PF approaches discussed in this paper. The economic valuation literature recognizes that such uncertainties create the conditions for *option values*, which arise from the difference between valuation under conditions of certainty and uncertainty (e.g., see Freeman, 2003; and Just, Hueth and Schmitz, 2004). The standard method recommended in the literature is to estimate this additional value separately, through various techniques to measure an *option price*, i.e. the amount of money that an individual will pay or must be compensated to be indifferent from the status quo condition of the ecosystem and the new, proposed condition.

However, in practice, estimating separate option prices for unknown ecological effects is very difficult. Determining the appropriate risk premium for vulnerable populations exposed to the irreversible ecological losses is also proving elusive. These are problems currently affecting all economic valuation methods of ecosystem services, and not just the production function approach. As one review of these studies concludes: "Given the imperfect knowledge of the way people value natural ecosystems, their goods and services, and our limited understanding of the underlying ecology and biogeochemistry of aquatic ecosystems, calculations of the value of the changes resulting from a policy intervention will always be approximate" (Heal *et al.,* 2005, p. 218).

Finally, recent attempts have been made to extend the production function approach to the ecosystem level through integrated ecological-economic modelling. This allows the ecosystem functioning and dynamics underlying the provision of ecological services to be modelled and can be used to value multiple rather than single services. For example, returning to the Thailand case study, it is well known that both coral reefs and sea grasses complement the role of mangroves in providing both the habitat-fishery and storm protection services. Thus full modelling of the integrated mangrove-coral reef-sea grass system could improve measurement of the benefits of both services. As we learn more about the important ecological and economic role played by such services, it may be relevant to develop multi-service production function modelling to understand more fully what values are lost when such integrated coastal and marine systems are disturbed or destroyed.

REFERENCES

Aksornkoae, S. & Tokrisna, R. 2004. Overview of shrimp farming and mangrove loss in Thailand. In: E.B. Barbier & S. Sathirathai (eds) *Shrimp Farming and Mangrove Loss in Thailand*, Edward Elgar, London. pp. 37-51.

Barbier, E.B. 1994. Valuing environmental functions: tropical wetlands. *Land Economics* 70: 155-173.

Barbier, E.B. 2000. Valuing the environment as input: applications to mangrove-fishery linkages. *Ecological Economics,* 35: 47-61.

Barbier, E.B. 2003. Habitat-fishery linkages and mangrove loss in Thailand. *Contemporary Economic Policy,* 21: 59-77.

Barbier, E.B. 2006. Valuing ecosystem services as productive inputs. Paper prepared for the 43rd Panel Meeting of *Economic Policy*, Vienna, Austria, 21-22 April 2006. Forthcoming in *Economic Policy*, January 2007.

Cameron, C.A. & Trivedi, P. 1998. *Regression Analysis of Count Data.* Cambridge University Press, Cambridge.

Carlsson, F., Frykblom, P. & Lijenstolpe, C. 2003. Valuing wetland attributes: an application of choice experiments. *Ecological Economics,* 47: 95-103.

Chong, J. 2005. Protective values of mangrove and coral ecosystems: a review of methods and evidence. IUCN, Gland, Switzerland.

Daily, G. (Ed) 1997. *Nature's Services: Societal Dependence on Natural Ecosystems.* Island Press, Washington, D.C.

De Groot, R.S., Wilson, M.A. & Boumans R.M.J. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics,* 41: 393-408.

Delgado, C., Wada, N., Rosegrant, M., Meijer, S. & Ahmed, M. 2003. Fish to 2020: Supply and Demand in Changing Global Markets. International Food Policy Research Institute, Washington, D.C.

FAO. 2000. Agriculture: Towards 2015/30. Technical Interim Report. Global Perspectives Unit. Food and Agricultural Organization of the United Nations, Rome.

FAO. 2005. *The State of World Fisheries and Aquaculture 2004.* Food and Agricultural Organization of the United Nations, Rome.

Freeman, A.M., III. 1991. Valuing environmental resources under alternative management regimes. *Ecological Economics,* 3: 247-256.

Freeman, A.M. III 2003. *The Measurement of Environmental and Resource Values: Theory and Methods*, 2nd ed. Resources for the Future, Washington, D.C.

Gollehon, N., Caswell, M., Ribaudo, M., Kellogg, M., Lander, C. & Letson, D. 2001. Confined Agricultural Production and Manure Nutrients. *Agricultural Information Bulletin* No. 771. Resource Economics Division, Economic Research Service, US Department of Agriculture, Washington, D.C.

Goldburg, R. & Naylor, R. 2005. Future seascapes, fishing, and fish farming. *Frontiers in Ecology and the Environment,* 3: 21-28.

Heal, G., Barbier, E., Boyle, K., Covich, A., Gloss, S., Hershner, C., Hoehn, J., Pringle, C., Polasky, S., Segerson, K., & Shrader-Frechette, K. 2005. *Valuing Ecosystem Services: Toward Better Environmental Decision Making*. The National Academies Press, Washington, D.C.

Jory, D.E. 1996. Marine shrimp farming in the Kingdom of Thailand: Part II. *Aquaculture Magazine,* July/August: 71-78.

Just, R.E., Hueth, D.L. & Schmitz, A. 2004. *The Welfare Economics of Public Policy: A Practical Approach to Project and Policy Evaluation.* Edward Elgar, Cheltenham, United Kingdom.

Krutilla, J.V. & Fisher, A.C. 1985. *The Economics of Natural Environments: Studies in the Valuation of Commodity and Amenity Resources.* Resources for the Future, Washington, D.C.

Mallin, M. & Cahoon, L. 2003. Industrialized animal production – a major source of nutrient and microbial pollution to aquatic ecosystems. *Population and Environment,* 24: 369-385.

Millennium Ecosystem Assessment. 2003. *Ecosystems and Human Well-being: A Framework for Assessment*. Island Press, Washington, D.C.

Naylor, R. & Drew, M. 1998. Valuing mangrove resources in Kosrae, Micronesia, *Environment and Development Economics,* 3: 471-490.

Naylor, R., Goldburg, R., Primavera, J., Kautsky, N., Beveridge, M., Clay, J., Folke, C., Lubchenco, J., Mooney, H. & Troll, M. 2000. Effect of aquaculture on world fish supplies. *Nature,* 405: 1017-1024.

Pagiola, S., von Ritter, K. & Bishop, J. 2004. How Much is an Ecosystem Worth? Assessing the Economic Value of Conservation. The World Bank, Washington, D.C.

Ready, R.C. 1995. Environmental valuation under uncertainty. In: D. W. Bromle (ed.) *The Handbook of Environmental Economics*, Blackwell Publishers, Cambridge, Mass. pp. 568-593.

Sathirathai, S. & Barbier, E. 2001. Valuing mangrove conservation, Southern Thailand. *Contemporary Economic Policy,* 19: 109-122.

Shabman, L.A. & Batie, S.S. 1978. Economic value of natural coastal wetlands: A critique. *Coastal Zone Management Journal,* 4: 231-247.

Tokrisna, R. 1999. An Economic Analysis of the Development of Marine Shrimp Culture in Thailand (Fisheries Circular), FAO, Rome.

Valiela, I., Bowen, J. & York, J. 2001. Mangrove forests: one of the world's threatened major tropical environments. *BioScience,* 51: 807-815.

Wilkie, M.L. & Fortuna, S. 2003. Status and trends in mangrove area extent worldwide. *Forest Resources Assessment Working Paper* No. 63. Forest Resources Division, Food and Agricultural Organization of the United Nations, Rome.

Winkelmann, R. 2003. *Econometric Analysis of Count Data*, 4th ed. Springer-Verlag, Berlin.