Valuing ecosystem services as productive inputs

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1. INTRODUCTION

Global concern over the disappearance of natural ecosystems and habitats has prompted policymakers to consider the 'value of ecosystem services' in environmental management decisions. These 'services' are broadly defined as 'the benefits people obtain from ecosystems' (Millennium Ecosystem Assessment, 2003, p. 53).

However, our current understanding of key ecological and economic relationships is sufficient to value only a handful of ecological services. An important objective of this paper is to explain and illustrate through numerical examples the difficulties faced in valuing natural ecosystems and their services, compared to ordinary economic or financial assets. Specifically, the paper addresses the following three questions:

1. What progress has been made in valuing ecological services for policy analysis?
2. What are the unique measurement issues that need to be overcome?
3. How can future progress improve upon the shortcomings in existing methods?

I am grateful to David Aadland, Carlo Favero, Geoff Heal, Omer Moav and three anonymous referees for helpful comments. The Managing Editor in charge of this paper was Paul Seabright.

Economic Policy January 2007 pp. 177–229 Printed in Great Britain

1.1. Key challenges and policy context

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. The diverse benefits of these functions include climate stability, maintenance of biodiversity and beneficial species, erosion control, flood mitigation, storm protection, groundwater recharge and pollution control (see Table 1 below).

One of the natural ecosystems that has seen extensive development and application of methods to value ecosystem services has been coastal wetlands. This paper focuses mainly on valuation approaches applied to these systems, and in particular their role as a nursery and breeding habitat for near-shore fisheries and in providing storm protection for coastal communities.

The paper employs a case study of mangrove ecosystems in Thailand to compare and contrast approaches to valuing habitat and storm protection services. Global mangrove area has been declining rapidly, with around 35% of the total area lost in the past two decades (Valiela *et al.*, 2001). Mangrove deforestation has been particularly prevalent in Thailand and other Asian countries. The main cause of global mangrove loss has been coastal economic development, especially aquaculture expansion (Barbier and Cox, 2003). Yet ecologists maintain that global mangrove loss is contributing to the decline of marine fisheries and leaving many coastal areas vulnerable to natural disasters. Concern about the deteriorating 'storm protection' service of mangroves reached new significance with the 26 December 2004 Asian tsunami that caused widespread devastation and loss of life in Thailand and other Indian Ocean countries.

The Thailand case study also illustrates the importance of valuing ecosystem services to policy choices. Because these services are 'non-marketed', their benefits are not considered in commercial development decisions. For example, the excessive mangrove deforestation occurring in Thailand and other countries is clearly related to the failure to measure explicitly the values of habitat and storm protection services of mangroves. Consequently, these benefits have been largely ignored in national land use policy decisions, and calls to improve protection of remaining mangrove forests and to enlist the support of local coastal communities through legal recognition of their *de facto* property rights over mangroves are unlikely to succeed in the face of coastal development pressures on these resources (Barbier and Sathirathai, 2004). Unless the value to local coastal communities of the ecosystem services provided by protected mangroves is estimated, it is difficult to convince policymakers in Thailand and other countries to consider alternative land use policies.

Thus, as the Thailand case study reveals, the challenge of valuing ecosystem services is also a policy challenge. Because the benefits of these services are important and should be taken into account in any future policy to manage coastal wetlands in Thailand and other countries, it is equally essential that economics continues to develop and improve existing methodologies to value ecological services.

1.2. Outline and main results

The paper makes three contributions. The first is to demonstrate that valuing ecological services as productive inputs is a viable methodology for policy analysis, and to illustrate the key steps through a detailed case study of mangroves in Thailand. The second contribution is to identify the measurement issues that make valuation of non-marketed ecosystem services a unique challenge, yet one that is important for many important policy decisions concerning the management of natural ecosystems. The third contribution of the paper is to show, using the examples of habitat and storm protection services, that improvements in methods for valuing these services can correct for some shortcomings and measurement errors, thus yielding more accurate valuation estimates. But even the preferred approaches display measurement weaknesses that need to be addressed in future developments of ecosystem valuation methodologies.

Section 2 discusses in more detail the importance of valuing ecosystem services, especially those arising from the regulatory and habitat functions to environmental decision-making. Section 3 reviews various methods for valuing these services. Because the benefits arising from ecological regulatory and habitat functions mainly support or protect valuable economic activities, the production function (PF) approach of valuing these benefits as environmental inputs is a promising methodology. However, the latter approach faces its own unique measurement issues. To illustrate the PF approach as well as its shortcomings, the section discusses recent advances using the examples of the habitat and storm protection services of coastal wetland ecosystems. Section 4 compares the application of the different methods to valuing mangroves in Thailand. The case study indicates the importance of considering the key ecological-economic linkages underlying each service in choosing the appropriate valuation approach, and how each approach influences the final valuation estimates. In the case of valuing the mangroves' habitat-fishery linkage, modelling the contribution of this linkage to growth in fish stocks over time appears to be a key consideration. The case study also demonstrates the advantages of the expected damage function approach as an alternative to the replacement cost method of valuing the storm protection service of coastal wetlands. Section 5 concludes the paper by discussing the key areas for further development in ecosystem valuation methodologies, such as incorporating the effects of irreversibilities, uncertainties and thresholds, and the application of integrated ecological-economic modelling to reflect multiple ecological services and their benefits. Although substantial progress has been made in valuing some ecosystem services, many difficulties still remain. Future progress in ecosystem valuation for policy analysis requires understanding the key flaws in existing methods that need correcting.

2. BACKGROUND: VALUATION OF ECOSYSTEM SERVICES

The rapid disappearance of many ecosystems has raised concerns about the loss of beneficial 'services'. This raises two important questions. What are ecosystem services, and why is it important to value these environmental flows?

2.1. Ecosystem services

Although in the current literature the term 'ecosystem services' lumps together a variety of 'benefits', economics normally classifies these benefits into 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, climate regulation, erosion control, etc.); and (iii) cultural benefits (e.g., spiritual and religious, heritage, etc.).[1] This paper focuses on methods to value a sub-set of the second category of ecosystem 'benefits' – the services arising from regulatory and habitat functions. Table 1 provides some examples of the links between regulatory and habitat functions and the resulting ecosystem benefits.

2.2. Valuing environmental assets

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.

However, what makes environmental assets special is that they give rise to particular measurement problems that are different for conventional economic or financial assets. This is especially the case for the benefits 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 *et al.*, 2004, p. 603). Although a natural ecosystem providing such beneficial services is unlikely to increase, it can be depleted, for example through habitat destruction, land conversion, pollution impacts and so

[1] See Daily (1997), De Groot *et al.* (2002) and Millennium Ecosystem Assessment (2003) for the various definitions of ecosystem services that are prevalent in the ecological literature.

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 run-off, 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 andnursery grounds	Maintenance of biodiversity Maintenance of beneficial species

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

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 is not revealed through market outcomes, then efficient management of such ecosystem services requires explicit methods to measure this social value (e.g., see Freeman, 2003; Just *et al.*, 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. The major cause of this disappearance is conversion of the land to other uses, degradation of the functioning and integrity of natural ecosystems through resource exploitation, pollution, and biodiversity loss, and habitat fragmentation (Millennium Ecosystem Assessment, 2003). 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' in development decisions as a consequence. Population and development pressures in many areas of the world result in increased land demand by economic activities, which mean that the opportunity cost of maintaining the land for natural ecosystems is rarely zero. Unless the benefits arising from ecosystem services are explicitly measured, or 'valued', then these non-marketed flows are likely to be ignored in land use decisions. Only the benefits of the 'marketed' outputs from economic activities, such as agricultural crops, urban housing and other commercial uses of land, will be taken into account, and as a consequence, excessive conversion of natural ecosystem areas for development will occur.

A further problem is the uncertainty over their future values of environmental assets. It is possible, for example, that the benefits of natural ecosystem services may increase in the future as more scientific information becomes available over time. In addition, if environmental assets are depleted irreversibly through economic development, their value will rise relative to the value of other economic assets (Krutilla and Fisher, 1985). Because ecosystems are in fixed supply, lack close substitutes and are difficult to restore, their beneficial services will decline as they are converted or degraded. As a result, the value of ecosystem services is likely to rise relative to other goods and services in the economy. This rising, but unknown, future scarcity value of ecosystem benefits implies an additional 'user cost' to any decision that leads to irreversible conversion today.

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 '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 *et al.*, 2004). However, it is difficult to implement such methods empirically, given the uncertainty over the future state of environmental assets and about the future preferences and income of individuals. The general conclusion from studies that attempt to allow for such uncertainties in valuing environmental assets 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).

3. VALUING THE ENVIRONMENT AS INPUT

Uncertainty and irreversible loss are important issues to consider in valuing ecosystem services. However, as emphasized by Heal *et al.* (2005), a more '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, then management of natural ecosystems and their services will not be efficient. The following section describes advances in developing the 'production function' approach, compared to other valuation methods, as a means to measuring the aggregate willingness to pay for the largely non-marketed benefits of ecosystem services.

3.1. Methods of valuing ecosystem services

Table 2 indicates various methods that can be used for valuing ecological services.[2] However, some approaches are limited to specific benefits. For example, the travel cost method is used principally for environmental values that enhance individuals' enjoyment of recreation and tourism, averting behaviour models are best applied to the health effects arising from pollution, and hedonic wage and property models are used primarily for assessing work-related 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 involve surveying individuals who benefit from an ecological service or range of services, and analysing the responses to measure individuals' willingness to pay for the service or services.

For example, choice experiments of wetland restoration in southern Sweden revealed that individuals' willingness to pay for the restoration increased if the result enhanced overall biodiversity but decreased if the restored wetlands were used mainly for the introduction of Swedish crayfish for recreational fishing (Carlsson *et al.*, 2003). In some cases, stated preference methods are used to elicit 'non-use values', that is, the additional 'existence' and 'bequest' values that individuals attach to ensuring that a well-functioning system will be preserved for future generations to enjoy. A contingent valuation study of mangrove-dependent coastal communities in Micronesia demonstrated 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).

However, 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 (Heal *et al.*, 2005). For many of the services arising from ecological regulatory and habitat

[2] It is beyond the scope of this paper to discuss all the valuation methods listed in Table 2. See Freeman (2003), Heal *et al.* (2005) and Pagiola *et al.* (2004) for more discussion of these various valuation methods and their application to valuing ecosystem goods and services.

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

[a] See Freeman (2003), Heal *et al.* (2005) and Pagiola *et al.* (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, such as many of the services listed in Table 1.

Source: Adapted from Heal *et al.* (2005, Table 4-2) and Table 1.

functions, 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 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. However, PF methods have their own measurement issues and limitations. These are also discussed further in the rest of this section, and illustrated using examples of key ecological services from coastal and estuarine wetlands.

3.2. The production function approach

Many of the beneficial services derived from regulatory and habitat functions are commonly classified by economists as indirect use values (Barbier, 1994). The benefits attributed to these services arise through their support or protection of activities that have directly measurable values (see Table 2). For example, coastal and estuarine wetlands, such as tropical mangroves and temperate marshlands, act as 'natural barriers' by preventing or mitigating storms and floods that could affect property and land values, agriculture, fishing and drinking supplies, as well as cause sickness and death. Similarly, coastal and estuarine wetlands may also provide a nursery and breeding habitat that supports the productivity of near-shore fisheries, which in turn may be valued for their commercial or recreational catch.

Because the benefits of these ecosystem services appear to enhance the productivity of economic activities, or protect them from possible damages, one possible method of measuring the aggregate willingness to pay for such services is to estimate their value as if they were a factor input in these productive activities. This is the essence of the PF valuation approaches, also called 'valuing the environment as input' (Barbier, 1994 and 2000; Freeman, 2003, ch. 9).[3]

The basic modelling approach underlying PF methods is similar to determining the additional value of a change in the supply of any factor input. 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 goods 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). The sum of consumer and producer surpluses in turn provides a measure of the willingness to pay for the improved ecosystem services.

[3] The concept of 'valuing' the environment as input is not new. Dose-response and change-in-productivity models, which have been used for some time, can be considered special cases of the PF approach in which the production responses to environmental quality changes are greatly simplified (Freeman, 1982).

An adaptation of the PF methodology is required in the case where ecological regulatory and habitat functions have a protective value, such as the storm protection and flood mitigation services provided by coastal wetlands. 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.

Although this paper focuses mainly on applications of the PF approach to coastal wetland ecosystems, as Table 2 indicates PF approaches are being increasingly employed for a diverse range of environmental quality impacts and ecosystem services. Some examples include maintenance of biodiversity and carbon sequestration in tropical forests (Boscolo and Vincent, 2003); nutrient reduction in the Baltic Sea (Gren *et al.*, 1997); pollination service of tropical forests for coffee production in Costa Rica (Ricketts *et al.*, 2004); tropical watershed protection services (Kaiser and Roumasset, 2002); groundwater recharge supporting irrigation farming in Nigeria (Acharya and Barbier, 2000); coral reef habitat support of marine fisheries in Kenya (Rodwell *et al.*, 2002); marine reserves acting to enhance the 'insurance value' of protecting commercial fish species in Sicily (Mardle *et al.*, 2004) and in the northeast cod fishery (Sumaila, 2002); and nutrient enrichment in the Black Sea affecting the balance between invasive and beneficial species (Knowler *et al.*, 2001).

3.3. Measurement issues for modelling habitat-fishery linkages

Applying PF methods to valuing ecosystem services has its own demands in terms of ecological and economic data. To highlight these additional measurement issues, this section draws on the example of valuing coastal wetlands as a nursery and breeding habitat for commercial near-shore fisheries.

First, application of the PF approach requires properly specifying the habitat-fishery PF model that links the physical effects of the change in this service to changes in market prices and quantities and ultimately to consumer and producer surpluses. As with many ecological services, it is difficult to measure directly changes in the habitat and nursery function of coastal wetlands. Instead, the standard approach adopted in coastal habitat-fishery PF models is to allow the wetland area to serve as a proxy for the productivity contribution of the nursery and habitat function (see Barbier, 2000 for further discussion). It is then relatively straightforward to estimate the impacts of the change in the coastal wetland area input on fishery catch, in terms of the marginal costs of fishery harvests and thus changes in consumer and producer surpluses.

Second, market conditions and regulatory policies for the marketed output will influence the values imputed to the environmental input (Freeman, 1991). For instance, the offshore fishery supported by coastal wetlands may be subject to open access. Under these conditions, profits in the fishery would be dissipated, and equilibrium prices would be equated to average and not marginal costs. As a consequence, there is no producer surplus, and the welfare impact of a change in wetland habitat is measured by the resulting change in consumer surplus only.

Third, if the ecological service supports a harvested natural resource system, such as a fishery, forestry or a wildlife population, then it may be necessary to model how changes in the stock or biological population may affect the future flow of benefits. If the natural resource stock effects are not considered significant, then the environmental changes can be modelled as impacting only current harvest, prices and consumer and producer surpluses. If the stock effects are significant, then a change in an ecological service will impact not only current but also future harvest and market outcomes. In the PF valuation literature, the first approach is referred to as a 'static model' of environmental change on a natural resource production system, whereas the second approach is referred to as a 'dynamic model' because it takes into account the intertemporal stock effects of the environmental change (Barbier, 2000; Freeman, 2003, ch. 9).

Finally, most natural ecosystems provide more than one beneficial service, and it may be important to model any trade-offs among these services as an ecosystem is altered or disturbed. Integrated economic-ecological modelling could capture more fully the ecosystem functioning and dynamics underlying the provision of key services, and can be used to value multiple services arising from natural ecosystems. For instance, integrated modelling of an entire wetland-coral reef-sea grass system could measure simultaneously the benefits of both the habitat-fishery linkage and the storm protection service provided by the system. Examples of such multi-service ecosystem modelling include analysis of salmon habitat restoration (Wu *et al.*, 2003); eutrophication of small shallow lakes (Carpenter *et al.*, 1999); changes in species diversity in a marine ecosystem (Finnoff and Tschirhart, 2003); and introduction of exotic trout species (Settle and Shogren, 2002).

To illustrate the first three of the above issues, I next explore two ways of measuring the welfare effects of an environmental change on a productive natural resource system with the example of the coastal habitat-fishery linkage. I will return to the issue of integrated ecological-economic modelling of multiple ecological services in Section 5.

3.3.1. Habitat-fishery linkages: static approaches. This section illustrates the use of a static model to value how a change in coastal wetland habitat area affects the market for commercially harvested fish. Many initial PF methods to value habitat-fishery linkages have relied on this static approach. For example, using data from the Lynne *et al.* (1981), Ellis and Fisher (1987) constructed such a model to value the support by Florida marshlands for Gulf Coast crab fisheries in terms of the resulting changes in consumer and producer surpluses from the marketed catch. Freeman (1991) then extended Ellis and Fisher's approach to show how the values imputed to the wetlands in the static model is influenced by whether or not the fishery is open access or optimally managed. Sathirathai and Barbier (2001) also used a static model of habitat-fishery linkages to value the role of mangroves in Thailand in supporting near-shore fisheries under both open access and optimally managed conditions.

As most near-shore fisheries are not optimally managed but open access, the following illustration of the static model of habitat-fishery linkages assumes that the fishery is open access. Any profits in the fishery will attract new entrants until all the profits disappear, and in equilibrium, the welfare change in coastal wetland is in terms of its impact on consumer surplus only.

As noted above, the general PF approach treats an ecological service, such as coastal wetland habitat, as an 'input' into the economic activity, and like any other input, its value can be equated with its impact on the productivity of any marketed output. More formally, if h is the marketed harvest of the fishery, then its production function can be denoted as:

$$h = h(E_i \ldots E_k, S) \tag{1}$$

The area of coastal wetlands, S, may therefore have a direct influence on the marketed fish catch, h, which is independent from the standard inputs of a commercial fishery, $E_i \ldots E_k$.

A standard assumption in most static habitat-fishery models is that the production function (1) takes the Cobb–Douglas form, $h = AE^aS^b$, where E is some aggregate measure of total effort in the off-shore fishery and S is coastal wetland habitat area. It follows that the optimal cost function of a cost-minimizing fishery is:

$$C^* = C(h, w, S) = wA^{-1/a}h^{1/a}S^{-b/a} \tag{2}$$

where w is the unit cost of effort. Assuming an iso-elastic market demand function, $P = p(h) = kh^{\eta}$, $\eta = 1/\varepsilon < 0$, then the market equilibrium for catch of the open access fishery occurs where the total revenues of the fishery just equals cost, or price equals average cost, i.e. $P = C^*/h$, which in this model becomes:

$$kh^{\eta} = wA^{-1/a}\mathrm{h}^{1-a/a}S^{-b/a} \tag{3}$$

which can be rearranged to yield the equilibrium level of fish harvest:

$$h = \left[\frac{w}{k}\right]^{a/\beta} A^{-1/\beta}S^{-b/\beta},\ \beta = (1+\eta)a - 1 \tag{4}$$

It follows from (4) that the marginal impact of a change in wetland habitat is:

$$\frac{dh}{dS} = -\frac{b}{\beta}\left[\frac{w}{k}\right]^{a/\beta} A^{-1/\beta}S^{-(b+\beta)/\beta} \tag{5}$$

The change in consumer surplus, CS, resulting from a change in equilibrium harvest levels (from h^0 to h^1) is:

$$\Delta CS = \int_{h^0}^{h^1} p(h)dh - [p^1h^1 - p^0h^0] = \frac{k[(h^1)^{\eta+1} - (h^0)^{\eta+1}]}{\eta + 1} - k[(h^1)^{\eta+1} - (h^0)^{\eta+1}]$$
$$= -\frac{\eta[p^1h^1 - p^0h^0]}{\eta + 1}. \qquad (6)$$

By utilizing (5) and (6) it is possible to estimate the new equilibrium harvest and price levels and thus the corresponding changes in consumer surplus associated with a change in coastal wetland area, for a given demand elasticity, γ.

Figure 1 is the diagrammatic representation of the welfare measure of a change in wetland area on an open access fishery corresponding to Equation (6). As shown in the figure, a change in wetland area that serves as a breeding ground and nursery for an open access fishery results in a shift in the average cost curve, AC, of the fishery. The welfare impact is the change in consumer surplus (area P*ABC).

3.3.2. Habitat-fishery linkages: dynamic approaches. If the stock effects of a change in coastal wetlands are significant, then valuing such changes in terms of the impacts on current harvest and market outcomes is a flawed approach. To overcome this shortcoming, a dynamic model of coastal habitat-fishery linkage incorporates the change in wetland area within a multi-period harvesting model of the fishery. The standard approach is to model the change in coastal wetland habitat as affecting the biological growth function of the fishery (Barbier, 2003). As a result, any value impacts of a change in this habitat-support function can be determined in terms of changes in the long-run equilibrium conditions of the fishery. Alternatively, the welfare analysis could be conducted in terms of the harvesting path that approaches this equilibrium or the path that is moving away from initial conditions in the fishery.

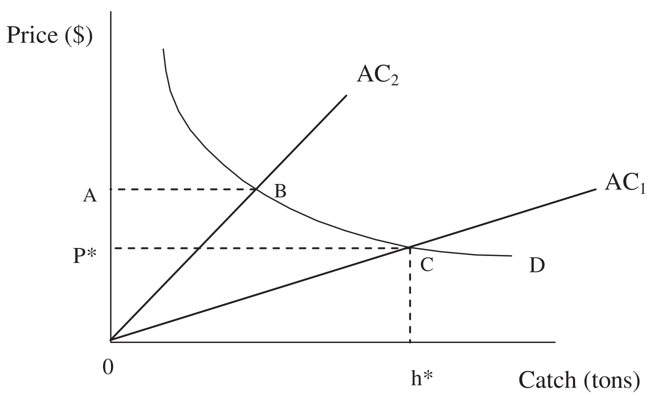

Figure 1. The economic value effects of increased wetland area on an open access fishery

Notes: AC: average cost; D: demand curve; P*: price per tonne; h*: fish catch in tonnes after change; P*ABC: change in consumer and producer surplus.

Source: Adapted from Freeman (1991).

Most attempts to value habitat-fishery linkages via a dynamic model that incorporates stock effects have assumed that the fishery affected by the habitat change is in a long-run equilibrium. Such a model has been applied, for example, in case studies of valuing habitat fishery linkages in Mexico (Barbier and Strand, 1998), Thailand (Barbier *et al.*, 2002; Barbier, 2003) and the United States (Swallow, 1994). Similar 'equilibrium' dynamic approaches have been used to model other coastal environmental changes, including the impacts of water quality on fisheries in the Chesapeake Bay (Kahn and Kemp, 1985; McConnell and Strand, 1989) and the effects of mangrove deforestation and shrimp larvae availability on aquaculture in Ecuador (Parks and Bonifaz, 1997).

However, valuing the change in coastal wetland habitat in terms of its impact on the long-run equilibrium of the fishery raises additional methodological issues. First, the assumption of prevailing steady state conditions is strong, and may not be a realistic representation of harvesting and biological growth conditions in the near-shore fisheries. Second, such an approach ignores both the convergence of stock and harvest to the steady state and the short-run dynamics associated with the impacts of the change in coastal habitat on the long-run equilibrium. The usual assumption is that this change will lead to an instantaneous adjustment of the system to a new steady state, but this in turn requires local stability conditions that may not be supported by the parameters of the model.

There are examples of pure fisheries models that assume that the dynamic system is not in equilibrium but is either on the approach to a steady state or is moving away from initial fixed conditions. The latter approach has proven particularly useful in the case of open access or regulated access fisheries (Bjørndal and Conrad, 1987; Homans and Wilen, 1997). The following model shows how this approach can be adopted here to the case of valuing a change in wetland habitat in terms of the dynamic path of an open access fishery.

Defining X_t as the stock of fish measured in biomass units, any net change in growth of this stock over time can be represented as:

$$X_t - X_{t-1} = F(X_{t-1}, S_{t-1}) - h(X_{t-1}, E_{t-1}), \frac{\partial^2 F}{\partial X_{t-1}^2} > 0, \frac{\partial F}{\partial S_{t-1}} > 0. \tag{7}$$

Thus, net expansion in the fish stock occurs as a result of biological growth in the current period, $F(X_{t-1}, S_{t-1})$, net of any harvesting, $h(X_{t-1}, E_{t-1})$, which is a function of the stock as well as fishing effort, E_{t-1}. The influence of the wetland habitat area, S_{t-1}, as a breeding ground and nursery habitat on growth of the fish stock is assumed to be positive, $\partial F/\partial S_{t-1} > 0$, as an increase in wetland area will mean more carrying capacity for the fishery and thus greater biological growth.

As before, it is assumed that the near-shore fishery is open access. The standard assumption for an open access fishery is that effort next period will adjust in response to the real profits made in the past period (Clark, 1976; Bjørndal and Conrad, 1987). Letting $p(h)$ represent landed fish price per unit harvested, w the unit cost of effort and $\phi > 0$ the adjustment coefficient, then the fishing effort adjustment equation is:

$$E_t - E_{t-1} = \phi[p(h_{t-1})h(X_{t-1}, E_{t-1}) - wE_{t-1}], \frac{\partial p(h_{t-1})}{\partial h_{t-1}} < 0. \tag{8}$$

Assume a conventional bioeconomic fishery model with biological growth characterized by a logistic function, $F(X_{t-1}, S_{t-1}) = rX_{t-1}[1 - X_{t-1}/K(S_{t-1})]$, and harvesting by a Schaefer production process, $h_t = qX_tE_t$, where q is a 'catchability' coefficient, r is the intrinsic growth rate and $K(S_t) = \alpha \ln S_t$, is the impact of coastal wetland area on carrying capacity, K, of the fishery. The market demand function for harvested fish is again assumed to be iso-elastic, i.e. $p(h) = kh^{\eta}$, $\eta = 1/\varepsilon < 0$. Substituting these expressions into (7) and (8) yields:

$$X_t = rX_{t-1}\left[1 - \frac{X_{t-1}}{\alpha \ln S_{t-1}}\right] - h_{t-1} + X_{t-1} \tag{9}$$

$$E_t = \phi R_{t-1} + (1 - \phi w)E_{t-1},\ R_{t-1} = kh_{t-1}^{1+\eta}. \tag{10}$$

Both X_t and E_t are predetermined, and so (9) and (10) can be estimated independently (see Homans and Wilen, 1997). Following Schnute (1977), define the catch per unit effort as $c_t = h_t/E_t = qX_t$. If X_t is predetermined so is c_t. Substituting the expression for catch per unit effort in (9) produces:

$$\frac{c_t - c_{t-1}}{c_{t-1}} = r - \frac{r}{q\alpha}\frac{c_{t-1}}{\ln S_{t-1}} - qE_{t-1}. \tag{11}$$

Thus Equations (10) and (11) can also be estimated independently to determine the biological and economic parameters of the model. For given initial effort, harvest and wetland data, both the effort and stock paths of the fishery can be determined for subsequent periods, and the consumer plus producer surplus can be estimated for each period. Alternative effort and stock paths can then be determined as wetland area changes in each period, and thus the resulting changes in consumer plus producer surplus in each period are the corresponding estimates of the welfare impacts of the coastal habitat change.[4]

3.4. Replacement cost and cost of treatment

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 as a valuation approach.[5] This method is usually invoked because of the lack of data for many services arising from natural ecosystems.

For example, the presence of a wetland may reduce the cost of municipal water treatment because the wetland system filters and removes pollutants. It is therefore

[4] As along its dynamic path the open access fishery is not in equilibrium, producer surpluses, or losses, are relevant for the welfare estimate of a change in coastal wetland habitat.

[5] Such an approach to approximating the benefits of a service by the cost of providing an alternative is not used exclusively in environmental valuation. For example, in the health economics literature this approach is referred to as 'cost of illness' (Dickie, 2003). This involves adding up the costs of treating a patient for an illness as the measure of the benefit to the patient of staying disease-free.

tempting to use the cost of an alternative treatment method, such as the building and operation of an industrial water treatment plant, to represent the value of the wetland's natural water treatment service. Such an approach does not measure directly the benefit derived from the wetland's waste treatment service; instead, the approach is estimating this benefit with the cost of providing the ecosystem service that people value. Herein lies the main problem with the replacement cost method: it is using 'costs' as a measure of economic 'benefit'. In economic terms, the implication is that the ratio of costs to benefit of an ecological service is always equal to one.

The problems posed by the replacement cost method are illustrated in Figure 2, in the case of waste water treatment service provided by an existing wetland ecosystem. The cost of the waste water treatment service provided by the wetlands is 'free' and thus corresponds to the horizontal axis, MC_S. Given the demand curve for water, Q_1 amount of water is consumed. However, if the wetland is destroyed the marginal cost of an alternative, human-built waste treatment facility is MC_H. Thus, the 'replacement cost' of using the treatment facility to provide Q_1 amount of water in the absence of the wetlands is the difference between the two supply curves, or area 0BDQ_1. However, this overestimates the benefit of having the wetlands provide the waste treatment service. The true benefit of this ecosystem service is the demand curve, or total willingness to pay, for Q_1 amount of water less the costs of providing it, or area 0ACQ_1.

For these reasons, economists consider that the replacement cost approach should be used with caution. Shabman and Batie (1978) suggested that this method can provide a reliable valuation estimation for an ecological service 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. In the absence of any information on benefits, and a decision has to be made to take some action, then treatment costs become a way of looking for a cost-effective action.

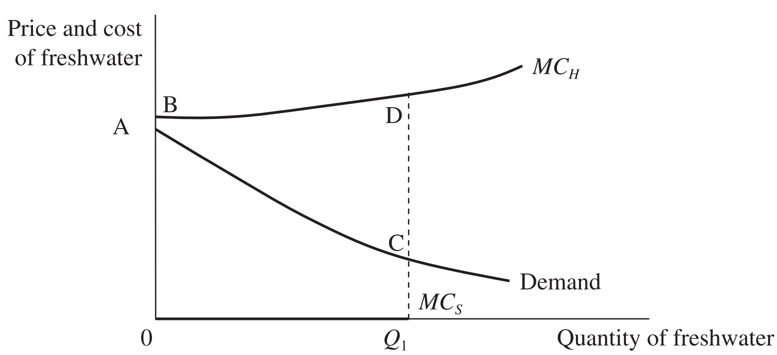

Figure 2. Replacement cost estimation of an ecosystem service

Source: Adapted from Ellis and Fisher (1987).

One of the best-known examples of a policy decision based on using the 'replacement cost' method to assess the value of an ecosystem service is the provision of clean drinking water by the Catskills Mountains for New York City (Heal *et al.*, 2005). In 1996, New York City faced a choice: either it could build water filtration systems to clean its water supply or the city could restore and protect the Catskill watersheds to ensure high-quality drinking water. Because estimates indicated that building and operating the filtration system would cost \$6–8 billion whereas protecting and restoring the watersheds would cost \$1–1.5 billion, New York chose to protect the Catskills. In this case, it was sufficient for the policy decision simply to demonstrate the cost-effectiveness of restoring and protecting the ecological integrity of the Catskills watersheds compared to the alternative of the human-constructed water filtration system. Thus, clearly this is an example where the criteria established by Shabman and Batie (1978) apply.

The main reason why economists have resorted to replacement cost approaches to valuing an ecosystem service, however, is that there is often a lack of data on the linkage between the initial ecological function, the processes and components of ecosystems that facilitate this function, and the eventual ecological service that benefits humans. The lack of such data makes it extremely difficult to construct reliable hypothetical scenarios through stated preference surveys and similar methods to elicit accurate responses from individuals about their willingness to pay for ecological services. As an illustration, in the Catskills case study, a stated preference survey may have elicited an estimate of the total willingness-to-pay by New York City residents for the amount of freshwater provided – for example, the total demand for freshwater Q_1 in Figure 2 – but it would have been very difficult to obtain a measure of the willingness-to-pay to avoid losses in the water treatment service that occur through *changes* in the land use in Catskills watershed that affect the free provision of this ecological service.

Similarly, as pointed out by Chong (2005), it is very difficult to use stated preference methods in tropical developing areas to assess the benefits to local communities of the storm protection service of mangrove systems. Although there is sufficient scientific evidence suggesting that such a service occurs, there is a lack of ecological data on how loss of mangroves in specific locations will affect their ability to provide storm protection to neighbouring communities. To date, 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 with constructed barriers that perform the same services (Chong, 2005). Unfortunately, such estimates not only make the classic error of estimating a 'benefit' by a 'cost' but also may yield unrealistically high estimates, given that removing all the mangroves and replacing them with constructed barriers is unlikely to be the least-cost alternative to providing storm prevention and flood mitigation services in coastal areas.

3.5. Expected damage function approach

For some ecological services, an alternative to employing replacement cost methods might be the *expected damage function* (EDF) approach.[6]

The EDF approach, which is a special category of 'valuing' the environment as 'input', 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, for example, as in the case of airline safety performance (Rose, 1990); highway fatalities (Michener and Tighe, 1992); drug safety (Olson, 2004); and studies of the incidence of diseases and accident rates (Cameron and Trivedi, 1998; Winkelmann, 2003). Here we show that the EDF approach can also be applied, under certain circumstances, to value ecological services that also reduce the probability and severity of economic damages.

Recall that one of the special features of many regulatory and habitat services of ecosystems is that they may protect nearby economic activities, property and even human lives from possible damages. As indicated in Table 1, such services include storm protection, flood mitigation, prevention of erosion and siltation, pollution control and maintenance of beneficial species. The EDF approach essentially 'values' these services through estimating how they mitigate damage costs.

The following example illustrates how the expected damage function (EDF) methodology can be applied to value the storm protection service provided by a coastal wetland, such as a marshland or mangrove ecosystem. The starting point is the standard 'compensating surplus' approach to valuing a quantity or quality change in a non-market environmental good or service (Freeman, 2003).

Assume that in a coastal region the local community owns all economic activity and property, which may be threatened by damage from periodic natural storm events. Assume also that the preferences of all households in the community are sufficiently identical so that it can be represented by a single household. Let $m(p^x, z, u^0)$ be the expenditure function of the representative household, that is, the minimum expenditure required by the household to reach utility level, u^0, given the vector of prices, p^x, for all market-purchased commodities consumed by the household, the expected number or incidence of storm events, z^0.

Suppose the expected incidence of storms rises from z^0 to z^1. The resulting expected damages to the property and economic livelihood of the household, $E[D(z)]$, translates into an exact measure of welfare loss through changes in the minimum expenditure function:

[6] The expected damage function approach predates many of the PF methods discussed so far, and has been used extensively to estimate the risk of health impacts from pollution (Freeman, 1982, chs. 5 and 9).

$$E[D(z)] = m(p^x, z^1, u^0) - m(p^x, z^0, u^0) = c(z) \tag{12}$$

where $c(z)$ is the compensating surplus. It is the minimum income compensation that the household requires to maintain it at the utility level u^0, despite the expected increase in damaging storm events. Alternatively, $c(z)$ can be viewed as the minimum income that the household needs to avoid the increase in expected storm damages.

However, the presence of coastal wetlands could mitigate the expected incidence of damaging storm events. Because of this storm protection service, the area of coastal wetlands, S, may have a direct effect on reducing the 'production' of natural disasters, in terms of their ability to inflict damages locally. Thus the 'production function' for the incidence of potentially damaging natural disasters can be represented as:

$$z = z(S),\ z' < 0,\ z'' > 0. \tag{13}$$

It follows from (12) and (13) that $\partial c(z)/\partial S = \partial E[D(z)]/\partial S < 0$. An increase in wetland area reduces expected storm damages and therefore also reduces the minimum income compensation needed to maintain the household at its original utility level. Alternatively, a loss in wetland area would increase expected storm damages and raises the minimum compensation required by the household to maintain its welfare. Thus, we can define the marginal willingness to pay, $W(S)$, for the protection services of the wetland in terms of the marginal impact of a change in wetland area on expected storm damages:

$$W(S) = -\frac{\partial E[D(z(S))]}{\partial S} = -E\left[\frac{\partial D}{\partial z} z'\right], W' < 0. \tag{14}$$

The 'marginal valuation function', $W(S)$, is analogous to the Hicksian compensated demand function for marketed goods. The minus sign on the right-hand sign of (14) allows this 'demand' function to be represented in the usual quadrant, and it has the normal downward-sloping property (see Figure 3). Although an increase in S reduces z and thus enables the household to avoid expected damages from storms, the additional value of this storm protection service to the household will fall as wetland area increases in size. This relationship should hold across all households in the coastal community. Consequently, as indicated in Figure 3, the marginal willingness to pay by the community for more storm protection declines with S.

The value of a non-marginal change in wetland area, from S_0 to S_1, can be measured as:

$$-\int_{S_0}^{S_1} W(S)dS = E[D(z(S))] = c(S). \tag{15}$$

If there is an increase in wetland area, then the value of this change is the total amount of expected damage costs avoided. If there is a reduction in wetland area, as shown in Figure 3, then the welfare loss is the total expected damages resulting from the increased incidence of storm events. As indicated in (15), in both instances the

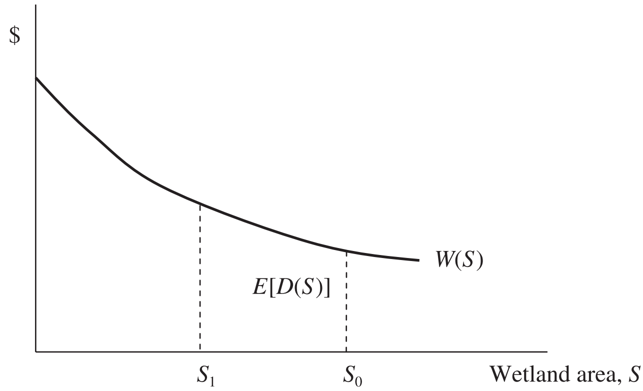

Figure 3. Expected damage costs from a loss of wetland area

valuation would be a compensation surplus measure of a change in the area of wetlands and the storm protection service that they provide.

As indicated in (14), an estimate of the marginal impact of a change in wetland area on expected storm damages has two components: the influence of wetland area on the expected incidence of economically damaging natural disaster events, z', and some measure of the additional economic damage incurred per event. Thus the right-hand expression in (14) can be estimated, provided that there are sufficient data on past storm events, and preferably across different coastal areas, and some estimate of the economic damages inflicted by each event. The most important step in the analysis is the first one, using the data on the incidence of past natural disasters and changes in wetland area in coastal areas to estimate $z(S)$. One way this analysis can be done is through employing *count data models*.

Count data models explain the number of times a particular event occurs over a given period. In economics, count data models have been used to explain a variety of phenomenon, such as explaining successful patents derived from firm R&D expenditures, accident rates, disease incidence, crime rates and recreational visits (Cameron and Trivedi, 1998; Greene, 2003, ch. 21; Winkelmann, 2003). Count data models could be used to estimate whether a change in the area of coastal wetlands, S, reduces the expected incidence of economically damaging storm events. The basic methodology for such an application of count data models is described further in the appendix.

However, applying the EDF method to estimating the storm protection value of coastal wetlands raises two additional measurement issues.

First, as the 2004 Asian tsunami and recent hurricanes in the United States have demonstrated, the risks to vulnerable populations living in coastal areas from the economic damages of storm events can be very large. This suggests that coastal populations will display a degree of risk aversion to such events, in the sense that they would like to see the least possible variance in expected storm damages. Applying standard techniques, such as the capital-asset pricing model, this implies in turn that there should be a 'risk premium' attached to the storm protection value of coastal wetlands that reduces the variance in expected economic damages from storm events (Hirshleifer and Riley, 1992).

Second, estimating how coastal wetlands affect the expected number of economic damaging events from the count data model and then multiplying the effect by the average economic damages across events could be misleading under some extreme circumstances. For instance, suppose a loss in wetland area is associated with a situation in which there is a change in the incidence of storms from one devastating storm to two relatively minor storms per year. The count data model would then be interpreted as not providing evidence against the null that the change in the wetland area increases expected storm damages. Clearly, there needs to be a robustness check on the count data model to ensure that such situations do not dominate the application of the EDF approach.

4. CASE STUDY OF MANGROVE ECOSYSTEMS IN THAILAND

This section illustrates the application of the PF approach and the EDF approach to valuation of ecological services with a case study of mangrove ecosystems in Thailand. The two services of interest are the provision of a breeding and nursery habitat for fisheries and the storm protection service of mangroves.

Both the dynamic and static PF approaches are used to estimate the value of the mangrove-fishery habitat service. The EDF approach to estimating the storm protection service of mangroves is contrasted with the replacement cost method.

4.1. Case study background

Many mangrove ecosystems, especially those in Asia, are threatened by rapid deforestation. At least 35% of global mangrove area has been lost in the past two decades; in Asia, 36% of mangrove area has been deforested, at the rate of 1.52% per year (Valiela *et al.*, 2001). Although many factors are behind global mangrove deforestation, a major cause is aquaculture expansion in coastal areas, especially the establishment of shrimp farms (Barbier and Cox, 2003). Aquaculture accounts for 52% of mangrove loss globally, with shrimp farming alone accounting for 38% of mangrove deforestation; in Asia, aquaculture contributes 58% to mangrove loss with shrimp farming accounting for 41% of total deforestation (Valiela *et al.*, 2001).

Mangrove deforestation has been particularly prevalent in Thailand. Some estimates suggest that over 1961–96 Thailand lost around 2050 km^2 of mangrove forests, or about 56% of the original area, mainly due to shrimp aquaculture and other coastal developments (Charuppat and Charuppat, 1997). Since 1975, 50–65% of Thailand's mangroves have been converted to shrimp farms (Aksornkoae and Tokrisna, 2004).

Figure 4 shows two long-run trend estimates of mangrove area in Thailand. In 1961, there were approximately 3700 km^2 of mangroves, which declined steadily to

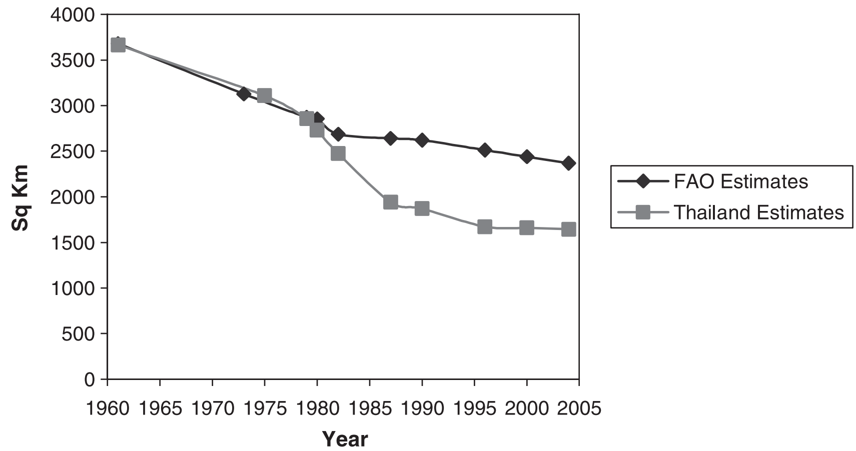

Figure 4. Mangrove area (km^2) in Thailand, 1961–2004

Notes: FAO estimates from FAO (2003). 2000 and 2004 data are estimated from 1990–2000 annual average mangrove loss of 18.0 km^2. Thailand estimates from various Royal Thailand Forestry Department sources reported in Aksornkoae and Tokrisna (2004). 2000 and 2004 data are estimated from 1993–96 annual average mangrove loss of 3.44 km^2.

Sources: Based on FAO (2003), Aksornkoae and Tokrisna (2004) and author's estimates.

around 2700 to 2900 km^2 by 1980. Since then, mangrove deforestation has continued, although there are disagreements over the rate of deforestation. For example, FAO estimates based on long-run trend rates suggest a slower rate of decline, and indicate that there may be almost 2400 km^2 of mangroves still remaining. However, estimates based on Thailand's Royal Forestry Department studies suggest that rapid shrimp farm expansion during the 1980s and early 1990s accelerated mangrove deforestation, and as a consequence, the area of mangroves in 2004 may be much lower, closer to 1,645 km^2.

Mangrove deforestation in Thailand has focused attention on the two principal services provided by mangrove ecosystems, their role as nursery and breeding habitats for offshore 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 $88 per hectare (ha), or approximately $823/ha in net present value terms over a 20-year period and with a 10% discount rate (Sathirathai and Barbier, 2001).

4.1.1. Breeding and nursery habitat for fisheries. An extensive literature in ecology has emphasized the role of coastal wetland habitats in supporting neighbouring marine fisheries (for a review, see Mitsch and Gosselink, 1993; World Conservation Monitoring Center; World Resources Institute 1996). Mangroves in Thailand also provide this important habitat service (Aksornkoae *et al.*, 2004).

Thailand's coastline is vast, stretching for 2815 km, of which 1878 km is on the Gulf of Thailand and 937 km on the Andaman Sea (Indian Ocean) (Kaosa-ard and Pednekar, 1998). Since 1972, the 3 km offshore coastal zone in southern Thailand has been reserved for small-scale, artisanal marine fisheries. The Gulf of Thailand is divided into four such major zones, and the Andaman Sea comprises a fifth zone.[7] The mangroves along these coastal zones are thought to provide breeding grounds and nurseries in support of several species of demersal fish and shellfish (mainly crab and shrimp) in Thailand's coastal waters.[8] The artisanal marine fisheries of the five major coastal zones of Thailand depend largely on shellfish but also some demersal fish. For example, in 1994 shrimp, crab, squid and cuttlefish alone accounted for 67% of all catch in the artisanal marine fisheries, and demersal fish accounted for 5.3% (Kaosa-ard and Pednekar, 1998).

The coastal artisanal fisheries of Thailand are characterized by classic open access conditions (Kaosa-ard and Pedneker, 1998; Wattana, 1998). Since the 1970s, there have been approximately 36 000–38 000 households engaged in small-scale fishing activities. Although there are 2500 fishing communities scattered over the 24 coastal provinces of Thailand, 90% of the artisanal fishing households are concentrated in communities spread along the Southern Gulf of Thailand and Andaman Sea coasts. While the number of households engaged in small-scale fishing has remained fairly stable since 1985, the use of motorized boats has increased by more than 30% (Wattana, 1998). Gill nets still remain the most common form of fishing gear used by artisanal fishers. Although a licence fee and permit are required for fishing in coastal waters, officials do not strictly enforce the law and users do not pay. Currently, there is no legislation for supporting community-based fishery management (Kaosa-ard and Pednekar, 1998).

4.1.2. Storm protection. The 26 December 2004 Indian Ocean tsunami disaster has focused attention on the role of natural barriers, such as mangroves, in protecting vulnerable coastlines and populations in the region from such storm events (UNEP, 2005; Wetlands International, 2005). Mangrove wetlands, which are found along sheltered tropical and subtropical shores and estuaries, are particularly valuable in minimizing damage to property and loss of human life by acting as a barrier against tropical storms, such as typhoons, cyclones, hurricanes and tsunamis (Chong, 2005; Massel *et al.*, 1999; Mazda *et al.*, 1997). Evidence from the 12 Indian Ocean countries affected by the tsunami disaster, including Thailand, suggests that those coastal areas

[7] The four Gulf of Thailand zones consist of the following coastal provinces: Trat, Chantaburi and Rayong (Zone 1); Chon Buri, Chachoengsao, Samut Parkakan, Samut Sakhon, Samut Songkhram, Phetchaburi, Prachaup Khiri Khan (Zone 2); Chumphon, Surat Thani, Nakhon Si Thammarat (Zone 3); and Songkhla, Patthani, Narathiwart (Zone 4). The fifth zone on the Indian Ocean (Andaman Sea) consists of the following coastal provinces: Ranong, Phangnga, Phuket, Krabi, Trang and Satun (Zone 5).

[8] Mangrove-dependent demersal fish include those belonging to the *Clupeidae*, *Chanidae*, *Ariidae*, *Pltosidae*, *Mugilidae*, *Lujanidae* and *Latidae* families. The shellfish include those belonging to the families of *Panaeidae* for shrimp and *Grapsidae*, *Ocypodidae* and *Portnidae* for crab.

that had dense and healthy mangrove forests suffered fewer losses and less damage to property than those areas in which mangroves had been degraded or converted to other land uses (Dahdouh-Guebas *et al.*, 2005; Harakunarak and Aksornkoae, 2005; Kathiresan and Rajendran, 2005; UNEP, 2005; Wetlands International, 2005).

In Thailand, the Asian tsunami affected all six coastal provinces along the Indian Ocean (Andaman Sea) coast: Krabi, Phang Nga, Phuket, Ranong, Satun and Trang. In Phang Nga, the most affected province, post-tsunami assessments suggest that large mangrove forests in the north and south of the province significantly mitigated the impact of the Tsunami. They suffered damage on their seaside fringe, but reduced the tidal wave energy, providing protection to the inland population (UNEP, 2005; Harakunarak and Aksornkoae, 2005). Similar results were reported for those shorelines in Ranong Province protected by dense and thriving mangrove forests. In contrast, damages were relatively extensive along the Indian Ocean coast where mangroves and other natural coastal barriers were removed or severely degraded (Harakunarak and Aksornkoae, 2005).

With the overwhelming evidence of the storm protection service provided by intact and healthy mangrove systems, since the tsunami disaster increased emphasis has been placed on replanting degraded and deforested mangrove areas in Asia as a means to bolstering coastal protection. For example, the Indonesian Minister for Forestry has announced plans to reforest 600 000 hectares of depleted mangrove forest throughout the nation over the next 5 years. The governments of Sri Lanka and Thailand have also stated publicly intentions to rehabilitate and replant mangrove areas (UNEP, 2005; Harakunarak and Aksornkoae, 2005).

Although the Asian tsunami has called attention to the storm protection service provided by mangroves, the benefits of this service extends to protection against many types of periodic coastal natural disaster events. As one post-tsunami assessment noted: 'It is important to recognize that any compromising of mangrove "protection function" is relevant to a wide variety of storm events, and not just tsunamis. Whereas the Indian Ocean area counted "only" 63 tsunamis between 1750 and 2004, there were more than three tropical cyclones per year in roughly the same area' (Dahdouh-Guebas *et al.*, 2005, pp. 445–6).

The EM-DAT International Disaster Database shows that the number of coastal natural disasters in Thailand has increased in both the frequency of occurrence and in the number of events per year (see Figure 5). Over 1975–87, Thailand experienced on average 0.54 coastal natural disasters per year, whereas between 1987–2004 the incidence increased to 1.83 disasters per year. Thus, a recent World Bank report identified the coastal and delta areas of Thailand as potentially high fatality (more than 1000 deaths per event) and other damage 'hotspots' at risk from storm surge events (Dilley *et al.*, 2005, pp. 101–3).

The EM-DAT database also calculates the economic damage incurred per event. Figure 6 plots the damages per coastal natural disaster in Thailand for 1975–2004. The 2004 Asian tsunami with estimated damages of US$240 million (1996 prices)

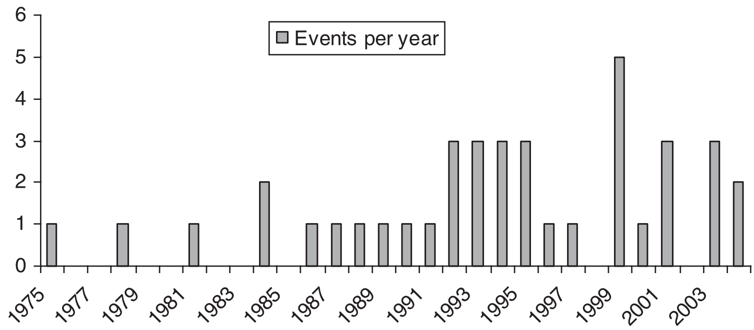

Figure 5. Coastal natural disasters in Thailand, 1975–2004

Notes: Over 1975–2004, coastal natural disasters included wave/surge (tsunami and tidal wave), wind storm (cyclone/typhoon and tropical storm) and flood (significant rise of water level in coastal region. In order for EM-DAT (2005) to record an event as a disaster, at least one or more of the following criteria must be fulfilled: 10 or more people reported killed; 100 people reported affected; declaration of a state of emergency; call for international assistance.

Source: EM-DAT (2005). EM-DAT: The OFDA/CRED International Disaster Database. www.em-dat.net – Université Catholique de Louvain, Brussels, Belgium.

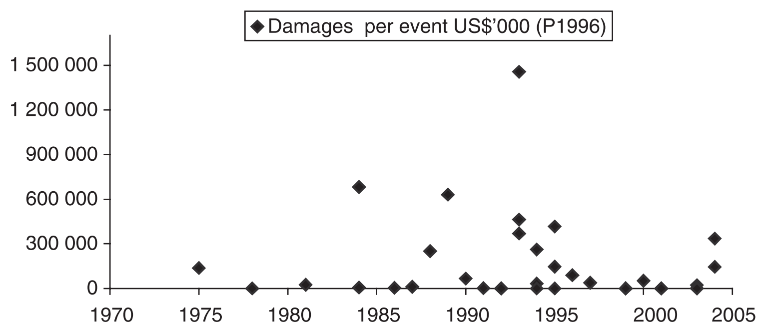

Figure 6. Real damages per coastal natural disaster in Thailand, 1975–2004

Notes: The EM-DAT (2005) estimate of the economic impact of a disaster usually consists of direct (e.g. damage to infrastructure, crops, housing) and indirect (e.g. loss of revenues, unemployment, market destabilization) consequences on the local economy. However, the estimate of 'zero' economic damages may indicate that no economic damages were recorded for an event. The estimates of economic damages are in thousands of US$ and converted to 1996 prices using Thailand's GDP deflator.

Source: EM-DAT (2005). EM-DAT: The OFDA/CRED International Disaster Database. www.em-dat.net – Université Catholique de Louvain, Brussels, Belgium.

was not the most damaging event to occur in Thailand. In fact, although the incidence of coastal damages has increased since 1987, in recent years the real damages per event has actually declined. For example, from 1979 to 1996, the economic damages per event were around US$190 million whereas from 1996 to 2004, real damages per event averaged US$61 million.

In sum, over the past two decades the rise in the number and frequency of coastal natural disasters in Thailand (Figure 5) and the simultaneous rapid decline in coastal mangrove systems over the same period (Figure 4) is likely to be more than a coincidence. Natural disasters occur when large numbers of economic assets are damaged or destroyed during a natural hazard event. Thus an increase in the incidence of coastal disasters is likely to have two sets of causes: the first is the natural hazards themselves – tsunamis and other storm surges, tidal waves, typhoons or cyclones, tropical storms and floods – but the second set is the increasing vulnerability of coastal populations, infrastructure and economic activities to being harmed or damaged by a hazard event.[9] The widespread loss of mangroves in coastal areas of Thailand may therefore have increased the vulnerability of these areas to more incidences of natural disasters.

4.2. Valuation of habitat-fishery linkage service

This subsection compares and contrasts the static and dynamic approaches outlined in Section 3.3 to valuing the habitat-fishery support service of mangroves in Thailand. As discussed above, in Thailand the near-shore artisanal fisheries supported by this ecological service are not optimally managed but largely open access.

To conduct the static production function analysis of the mangrove-fishery linkage, the methodology of Section 3.3.1 is applied to the same shellfish, demersal fishery and mangrove data over 1983–96 as in Barbier (2003). These comprise pooled time-series and cross-sectional data over the 1983–96 period for Thailand's artisanal and shellfish fisheries, as well as the extent of mangrove area, corresponding to the five coastal zones along the Gulf and Thailand and Indian Ocean (Andaman Sea). Evidence from domestic fish markets in Thailand suggest that the demand for fish is fairly inelastic, and an elasticity of -0.5 was assumed for the iso-elastic market demand function. Thus the static analysis calculation of the marginal impact of a change in wetland area in Equation (5) requires specifying the unknown parameters of the Cobb–Douglas production function for the fishery, $h = AE^aS^b$. Section A1 in the appendix explains the approach used to estimate the unknown parameters (A, a, b) of the log-linear version of the Cobb–Douglas production function and reports the resulting preferred estimations (see Table A1). Using these results in Equations (5) and (6) allows calculation of the welfare impacts of mangrove deforestation on Thailand's two artisanal fisheries. The results are depicted in Table 3, which displays both the point estimates and the 95% confidence bounds on these estimates through use of the standard errors. All price and cost data for the fisheries used in the welfare analysis are in 1996 real terms.

[9] This view that natural disasters should not be viewed solely as 'acts of God' but clearly have an important anthropogenic component to their cause is reflected in much of the current expert opinion on natural disaster management. This is summarized succinctly by Dilley *et al.* (2005, p. 115): 'Hazards are not the cause of disasters. By definition, disasters involve large human or economic losses. Hazard events that occur in unpopulated areas and are not associated with losses do not constitute disasters. Losses are created not only by hazards, therefore, but also by the intrinsic characteristics of the exposed infrastructure, land uses, and economic activities that cause them to be damaged or destroyed when a hazard strikes. The socioeconomic contribution to disaster causality is potentially a source of disaster reduction. Disaster losses can be reduced by reducing exposure or vulnerability to the hazards present in a given area.'

Table 3. Valuation of mangrove-fishery linkage service, Thailand, 1996–2004 (US$)

Production function approach	Average annual mangrove loss	
	FAO (18.0 km^2)[a]	Thailand (3.44 km^2)[b]
Static analysis:		
Annual welfare loss	99 004 (12 704–814 504)	18 884 (2425–154 307)
Net present value	570 167	108 756
(10% discount rate)	(55 331–4 690 750)	(10 563–888 657)
Net present value	527 519	100 621
(12% discount rate)	(52 233–4 339 883)	(9972–822 186)
Net present value	472 407	90 108
(15% discount rate)	(48 080–3 886 476)	(9179–736 288)
Dynamic analysis:		
Net present value	1 980 128	373 404
(10% discount rate)	(403 899–2 390,728)	(164 506–691 573)
Net present value	1 760 374	331 995
(12% discount rate)	(357 462–2 104 176)	(147 571–614 058)
Net present value	1 484 461	279 999
(15% discount rate)	(299 411–1 747 117)	(126 178–516 691)

Notes: All valuations are based on mangrove-fishery linkage impacts on artisanal shellfish and demersal fisheries in Thailand at 1996 prices. The demand elasticity for fish is assumed to be −0.5. Figures in parentheses represent upper and lower bound welfare estimates based on the standard errors of the estimated parameters in each model (see Section A1 in the appendix).

[a] FAO estimates from FAO (2003). 2000 and 2004 data are estimated from 1990–2000 annul average mangrove loss of 18.0 km^2.

[b] Thailand estimates from various Royal Thailand Forestry Department sources reported in Aksornkoae and Tokrisna (2004). 2000 and 2004 data are estimated from 1993–96 annual average mangrove loss of 3.44 km^2.

Sources: Author's calculations.

As Figure 4 shows, there are two different estimates of the 1996–2004 annual mangrove deforestation rates in Thailand, namely the FAO estimate of 18.0 km^2 and the Royal Thai Forestry Department estimate of 3.44 km^2. For the welfare impacts arising from the FAO estimates of annual average mangrove deforestation rates in Thailand over 1996–2004, the static analysis suggests that the annual loss in the habitat-fishery support service is around US$99 000 ($13 000 to 815 000 with 95% confidence). The net present value of these losses over the entire period is between US$0.47 and 0.57 million ($48 000 to 4.7 million with 95% confidence). For the much lower Thailand deforestation estimates, the annual welfare loss is just under $19 000 ($2400 to 154 000 with 95% confidence) and the net present value of these losses over the 1996–2004 period is US$90 000 to 108 000 ($9000 to 0.9 million with 95% confidence).

Following the methodology of Section 3.3.2, we can also apply a dynamic production function model to mangrove-fishery linkages in Thailand. As explained in the section, this approach involves estimating the parameters of the dynamic mangrove-fishery model, and then using these parameters to simulate the dynamic path of the fishery and the corresponding consumer and surplus changes resulting from mangrove deforestation. Because there are no data on the biomass stock, X_t, for

Thailand's near-shore fisheries, the appropriate dynamic model is the version indicating the change over time in fishing effort, E_t, and catch per unit effort, c_t, i.e. Equations (10) and (11). To compare with the static analysis, we use the same shellfish, demersal fisheries and mangrove data, as well as assume the same iso-elastic demand, from Barbier (2003) to estimate Equations (10) and (11) (see Section A2 in the appendix). For example, the estimated parameters in the appendix correspond to the following parameters of the dynamic production function model: $b_0 = r$, $b_1 = -r/q\alpha$, $b_2 = -q$, $b_0/(b_1 * b_2) = \alpha$, $a_1 = \phi$, $a_2 = 1-\phi w$ and $-(a_2 - 1)/a_1 = w$. These estimated parameters are then employed to simulate the dynamic effort and stock paths (9) and (10) of each fishery, starting from an initial level of effort, catch per unit effort and mangrove area, and assuming a constant elasticity of demand of -0.5.[10] By using 1996 data as the initial starting point in the simulation, i.e. for X_0, E_0, S_0 and h_0, the dynamic paths yield effort, stock and harvest for each subsequent year from 1996–2004.

In the base case dynamic simulation, mangrove area is held constant at 1996 levels. Two alternative paths for stock, effort and thus harvest are then also simulated, corresponding to the two different estimates of the 1996–2004 annual mangrove deforestation rates in Thailand, namely the FAO estimate of 18.0 km^2 and the Royal Thai Forestry Department estimate of 3.44 km^2 respectively. The resulting changes in consumer plus producer surpluses in each year over 1996–2004, between each deforestation simulation and the base case, provide the estimates of the welfare impacts of the decline in the mangrove-fishery support service. That is, the changes in consumer and producer surplus resulting from mangrove deforestation in each subsequent year of the simulation are discounted to obtain a net present value estimate of the resulting welfare loss. As in the static analysis, the discount rate is varied from 10% to 15% (see Table 3). The standard errors for the parameters of the model estimated from Equations (10) and (11) were also used to construct both lower and upper confidence bounds on the simulation paths, and thus also on the welfare estimates of the impacts of deforestation on the mangrove-fishery linkage.

The results for the dynamic mangrove-fishery linkage analysis are also depicted in Table 3, which indicates the welfare calculations associated with both the FAO and Thailand deforestation estimates over 1996–2004. The table reports calculations arising from the simulations based on the point estimates of the parameters of the dynamic mangrove-fishery model. The ranges of values indicated in parentheses for the dynamic analysis represent the lower and upper bound confidence intervals

[10] Although there are no reliable stock data for Thailand's near-shore fisheries, the Schaefer harvesting function, $h_t = qE_tX_t$, assumed in the model allows stock to be determined from catch per unit effort for a given estimated parameter q. That is, $X_t = c_t/q$, where $c_t = h_t/E_t$. See Schnute (1977) for further details. The procedure employed here is to use the known harvest and effort levels, as well as the estimated parameter q, for each fishery in the initial year 1996 to estimate the initial unknown stock level, X_0. Equations (9) and (10) were then used to simulate the dynamic path for X_t and E_t in the subsequent years (1997–2004), as well as the subsequent harvest, $h_t = qE_t X_t$. The dynamic simulation approach employed here is standard for an open access fishery model (see Bjørndal and Conrad, 1987; Clark, 1976; Homans and Wilen, 1997).

derived from the standard errors of the estimated model parameters (see Section A2 in the appendix). If the FAO estimate of mangrove deforestation over 1996–2004 is used, then the net present value of the welfare loss ranges from around US$1.5 to 2.0 million ($0.3 to 2.4 million in the upper and lower bound simulation estimates). In contrast, the lower Thailand deforestation estimation for 1996–2004 suggests that the net present value welfare loss from reduced mangrove support for fisheries is around US$0.28 to 0.37 million ($0.13 to 0.69 million in the upper and lower bound simulation estimates).

The welfare estimates in Table 3 indicate that the losses in the habitat-fishery support service caused by mangrove deforestation in Thailand over 1996–2004 are around three times greater for the dynamic production function approach compared to the static analysis. In addition, the confidence bounds on the welfare estimates produced with the static analysis are significantly larger, suggesting that the static approach yields much more variable estimates of the welfare losses. Given the disparity in estimates between the two approaches, a legitimate question to ask is whether or not one approach should be preferred to the other in valuing habitat-fishery linkages.

It has been argued in the literature that, on the methodological grounds, the 'dynamic' PF approach is more appropriate for valuing how coastal wetland habitats support offshore fisheries because this service implies that fish populations are more likely to be affected over time (Barbier, 2000). If this is the case, then the environmental 'input' of mangroves serving as breeding and nursery habitat for near-shore fisheries should be modelled as part of the growth function of the fish stock. In contrast, the static analysis, by definition, ignores stock effects and focuses exclusively on the impact of changes in mangrove area on fishing effort and costs in the same period in which the habitat service changes. The comparison of the dynamic and static analysis in the Thailand case study of mangrove-fishery linkages confirms that, by incorporating explicitly the multi-period stock effects resulting from mangrove loss, the dynamic model produces much larger estimates for the value of changes in the habitat-fishery support service. Since in this case study at least these stock effects appear to be considerable, then they are clearly an important component of the impacts of mangrove deforestation on the habitat-fishery service in Thailand.

In sum, the Thailand case study suggest caution in using the static analysis in preference to the dynamic production function approach in valuing the ecological service of coastal wetlands as breeding and nursery habitat for offshore fisheries. As Table 3 indicates, the static approach could underestimate the value of this service as well as yield more variable estimates. This may prove misleading for policy analysis, particularly when considering options to preserve as opposed to convert coastal wetlands. Certainly, the perception among coastal fishing communities throughout Thailand is that the habitat-fishery service of mangroves is vital, and local fishers in these communities have reported substantial losses in coastal fish stocks and yields, which they attribute to recent deforestation (Aksornkoae *et al.*, 2004; Sathirathai and Barbier, 2001).

4.3. Valuation of storm protection service

To date, the most prevalent method of valuing the storm protection service provided by coastal wetlands is the replacement cost approach (Chong, 2005). This paper has suggested the use of an alternative methodology, the EDF approach. The purpose of the following subsection is to compare and contrast both approaches, using the Thailand case study.

Sathirathai and Barbier (2001) employed the replacement cost method to estimate the value of coastal protection and stabilization provided by mangroves in southern Thailand. The same approach and data will be employed here. According to the Harbor Department of the Royal Thai Ministry of Communications and Transport, the unit cost of constructing artificial breakwaters to prevent coastal erosion and damages from storm surges is estimated to be US$1011 (in 1996 prices) per metre of coastline. Based on this estimate, the authors calculate the equivalent cost of protecting the shoreline with a 75-metre width stand of mangrove is approximately US$13.48 per m^2, or US$134 801 per ha (1996 prices). Over a 20-year period and assuming a 10% discount rate, the annualized value of this cost amounts to $14 169 per ha. This is the 'replacement cost' value of the storm protection function per ha of mangrove.

The analysis for this paper uses this replacement cost value to calculate the annual and net present value welfare losses associated with the two mangrove deforestation estimates for Thailand over 1996–2004. The results are depicted in Table 4.

For the FAO mangrove deforestation estimate of 18.0 km^2 per year over 1996–2004, the annual welfare loss in storm protection service is around US$25.5 million, and the net present value of this loss over the entire period ranges from US$121.7 to 146.9 million. For the Thailand deforestation estimation of 3.4 km^2 per year, the annual welfare loss in storm protection is about US$4.9 million, and the net present value of this loss over the entire period ranges from US$23.2 to 28 million.

Section 3.5 describes the methodology for the EDF approach to estimating the value of the storm protection service of coastal wetlands such as mangroves. As emphasized in the appendix, the key step to this approach is to estimate the influence of changes in coastal wetland area on the expected incidence of economically damaging natural disaster events. The application of the EDF approach here employs a count data model for this purpose. The details of the estimation are contained in Section A3 of the appendix.

The analysis for Thailand over 1979–96 shows that loss of mangrove area in Thailand increases the expected number of economically damaging natural disasters affecting coastal provinces. Using this estimated 'marginal effect' (–0.00308), it is possible to estimate the resulting impact on expected damages of natural coastal disasters. For example, EM-DAT (2005) data show that over 1979–96 the estimated real economic damages per coastal event per year in Thailand averaged around US$189.9 million (1996 prices). This suggests that the marginal effect of a one-km^2

Table 4. Valuation of storm protection service, Thailand, 1996–2004 (US$)

Valuation approach	Average annual mangrove loss	
	FAO (18.0 km^2)[a]	Thailand (3.44 km^2)[b]
Replacement cost method:[c]		
Annual welfare loss	25 504 821	4 869 720
Net present value (10% discount rate)	146 882 870	28 044 836
Net present value (12% discount rate)	135 896 056	25 947 087
Net present value (15% discount rate)	121 698 392	23 236 280
Expected damage function approach:		
Annual welfare loss	3 382 169 (2 341 686–5 797 339)	645 769 (447 106–1 106 905)
Net present value (10% discount rate)	19 477 994 (13 485 827–33 387 014)	3 718 998 (2 574 894–6 374 694)
Net present value (12% discount rate)	18 021 043 (12 477 089–30 889 671)	3 440 818 (2 382 292–5 897 868)
Net present value (15% discount rate)	16 138 305 (11 173 553–27 662 490)	3 081 340 (2 133 404–5 281 692)

Notes: Figures in parentheses represent upper and lower bound welfare estimates based on the 95% confidence interval for the estimated coefficients in the model (see Section A3 in the appendix).
[a] FAO estimates from FAO (2003). 2000 and 2004 data are estimated from 1990–2000 annual average mangrove loss of 18.0 km^2.
[b] Thailand estimates from various Royal Thailand Forestry Department sources reported in Aksornkoae and Tokrisna (2004). 2000 and 2004 data are estimated from 1993–96 annual average mangrove loss of 3.44 km^2
[c] Re-calculated based on Sathirathai and Barbier (2001).

Sources: Author's calculations.

loss of mangrove area is an increase in expected storm damages of about US$585 000 per km^2. In Table 4, this latter calculation is combined with the FAO and Thailand estimates of the average annual rates of deforestation to compute the welfare losses in storm protection service for Thailand over 1996–2004. The table shows the welfare calculations based both on the point estimates of the count data regression and on using the standard errors to construct 95% confidence bounds on these estimates.

Table 4 shows that, for the FAO mangrove deforestation estimate of 18.0 km^2 per year over 1996–2004, the EDF approach estimates the annual welfare loss in storm protection service to be around US$3.4 million ($2.3 to 5.8 million with 95% confidence), and the net present value of this loss over the entire period ranges from US$16.1 to 19.5 million ($11.2 to 33.4 million with 95% confidence). For the Thailand deforestation estimation of 3.4 km^2 per year, the annual welfare loss in storm protection is over US$0.65 million ($0.45 to 1.1 million with 95% confidence), and the net present value of this loss over the entire period ranges from US$3.1 to 3.7 million ($2.1 to 6.4 million with 95% confidence).

Comparing the EDF approach and the replacement cost method of estimating the welfare impacts of a loss of the storm protection service due to mangrove deforestation confirms that the replacement cost method tends to produce extremely high estimates – almost 4 times greater than even the largest upper-bound estimate calculated using the EDF approach. This suggests that the replacement cost method should be used with caution, and when data are available, the EDF approach may provide more reliable values of the storm protection service of coastal wetlands.

4.4. Land use policy implications

Valuation of the ecosystem services provided by mangroves are important for two land use policy decisions in Thailand. First, although declining in recent years, conversion of remaining mangroves to shrimp farm ponds and other commercial coastal developments continues to be a major threat to Thailand's remaining mangrove areas. Second, since the December 2004 tsunami disaster, there is now considerable interest in rehabilitating and restoring mangrove ecosystems as 'natural barriers' to future coastal storm events.

To illustrate how improved and more accurate valuation of ecosystems can help inform these two policy decisions, Table 5 compares the per hectare net returns to shrimp farming, the costs of mangrove rehabilitation and the value of mangrove services. All land uses are assumed to be instigated over 1996–2004 and are valued in 1996 US dollars. The net economic returns to shrimp farming are based on non-declining yields over a 5-year period of investment, with the pond abandoned in subsequent years (Sathirathai and Barbier, 2001). These returns to shrimp aquaculture are estimated to be \$1078 to \$1220 per ha. In comparison, the costs rehabilitating mangrove ecosystems on land that has been converted to shrimp farms and then abandoned are \$8812 to \$9318 per ha. Thus valuing the goods and services of mangrove ecosystems can help to address two important policy questions: Do the net economic returns to shrimp farming justify further mangrove conversion to this economic activity, and is it worth investing in mangrove replanting and ecosystem rehabilitation in abandoned shrimp farm areas?

As indicated in Table 5, if the older methods of valuing habitat-fishery linkages with the static approach and storm protection with the replacement cost method are employed, then mangrove ecosystem benefits are considerably higher than the net economic returns to shrimp farming and the costs of replanting and rehabilitating mangroves in abandoned farm areas. However, the static analysis undervalues the habitat-fishery linkage of mangroves whereas the replacement cost method over-inflates storm protection. The replacement cost method estimates storm protection at \$67 610 to 81 602 per ha, which is 99% of the value of all mangrove ecosystem benefits. In contrast, the net income to local coastal communities from collected forest products and the value of habitat-fishery linkages total to only \$730 to \$881 per ha, which suggests that these two benefits of mangroves are insufficient on their own to justify either halting conversion to shrimp farms or replanting and rehabilitating these ecosystems on abandoned pond land.

If improved methods of valuing habitat-fishery linkages by the dynamic approach and storm protection by the expected damage function method are employed, then

Table 5. Comparison of land use values per hectare, Thailand, 1996–2004 (US$)

Land use	Net present value per hectare (10–15% discount rate)	
Shrimp farming:[a]		
Net economic returns	1078–1220	
Mangrove ecosystem rehabilitation:[b]		
Total cost	8812–9318	
Ecosystem goods and services	*Older methods*	*Improved methods*
Net income from collected forest products[c]	484–584	484–584
Habitat-fishery linkage	246–297[d]	708–987[e]
Storm protection service	67 610–81 602[f]	8966–10 821[g]
Total	68 341–82 484	10 158–12 392

[a] Based on Sathirathai and Barbier (2001), updated to 1996 US$.
[b] Based on costs of rehabilitating abandoned shrimp farms, replanting mangrove forests and maintaining and protecting mangrove seedlings. From Sathirathai and Barbier (2001), updated to 1996 US$.
[c] Based on Sathirathai and Barbier (2001), updated to 1996 US$.
[d] Based on marginal value per ha based on the static analysis of this study (see Section 4.2).
[e] Based on average value per hectare over 1996–2004 based on the dynamic analysis of this study and assuming the estimated Thailand deforestation rate of 3.44 sq km per year (see Section 4.2).
[f] Based on average value per hectare of replacement cost method of this study (see Section 4.3).
[g] Based on marginal value per hectare of expected damage function approach of this study (see Section 4.3).
Sources: Author's calculations.

the outcome is somewhat different. Although the total value of mangrove ecosystem services is lowered to $10 158 to $12 392 per ha, it still exceeds the net economic returns to shrimp farming. Storm protection service is still the largest benefit of mangroves, but it no longer dominates the land use value comparison. The net income to local coastal communities from collected forest products and the value of habitat-fishery linkages total to $1192 to $1571 per ha, which now are greater than the net economic returns to shrimp farming. The value of the storm protection, however, is critical to the decision as to whether or not to replant and rehabilitate mangrove ecosystems in abandoned pond areas. As shown in Table 5, storm protection benefits make mangrove rehabilitation an economically feasible land use option.

5. CONCLUSIONS

The case study of valuing mangroves in Thailand illustrates the potential use of the PF approach to modelling key ecological regulatory and habitat services. The study also indicates the importance of choosing the appropriate PF method for modelling the key ecological-economic linkages underlying each service.

For example, the case study confirms that, if coastal wetlands such as mangroves serve as a breeding and nursery habitat for a variety of near-shore fisheries, then it seems more appropriate to model this environmental input as part of the growth function of the fish stock. In comparison, not accounting for the stock effects of a change in coastal nursery and breeding grounds may lead to an underestimation of the value of this habitat-fishery linkage. The case study also illustrates how the EDF approach can be applied to valuing the storm protection service provided by mangroves, and demonstrates why this method should be preferred to the less-reliable replacement cost method, which has been used extensively in the literature to date (Chong, 2005).

The case study also points to some important policy implications for Thailand. In recent decades, considerable mangrove deforestation has taken place in Thailand, mainly as a result of shrimp farm expansion and other coastal economic developments (see Figure 4). Over this period, mangrove conversion for these development activities was systematically encouraged by government land use policies (Aksornkoae and Tokrisna, 2004; Barbier, 2003; Sathirathai and Barbier, 2001). Such policies were designed without consideration of the value of the ecological services provided by mangroves, such as their habitat support for coastal fisheries and storm protection. The case study of valuing these ecological services for Thailand illustrates that their benefits are significant, and should certainly not be ignored in future mangrove land management decisions.

The case study applications in this paper of valuing coastal storm protection and habitat services have policy implications beyond Thailand as well. Even before Hurricanes Katrina and Rita devastated the central Gulf Coast of the United States in 2005, the US Army Corps of Engineers had proposed a $1.1 billion multi-year programme to slow the rate of wetland loss and restore some wetlands in coastal Louisiana. In the aftermath of these hurricanes, the US Congress is now considering expanding the programme substantially to a $14 billion restoration effort (Zinn, 2005). As noted in Section 4, in the wake of the 2004 Asian tsunami, mangrove restoration projects for enhanced coastal protection are underway in many countries throughout the region. International donor groups are also supporting mangrove restoration projects in Asia, especially in countries and regions devastated by the tsunami (Check, 2005). In addition, there is mounting scientific evidence that near-shore fisheries throughout the world are undergoing rapid decline, with loss of coastal habitat and nursery grounds for these fisheries a contributory cause (Jackson *et al.*, 2001; Myers and Worm, 2003). Valuing the storm protection and habitat services of coastal wetlands, as illustrated by the Thailand case study in this paper, can therefore play a vital role in current and future debates about the state of coastal ecosystems worldwide and the assessment of the costs and benefits of restoring these vital ecosystems.

Thus, valuing the non-market benefits of ecological regulatory and habitat services is becoming increasingly important in assisting policymakers to manage critical environmental assets. 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. 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 *et al.*, 2004). The standard approach recommended in the literature is to estimate this additional value separately, through various techniques to measure an *option price*, that is, 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 and 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, Section 3 noted recent attempts 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 ecosystem modelling to understand more fully what values are lost when such integrated coastal and marine systems are disturbed or destroyed.

Discussion

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The objective of the paper is to apply a production function (and expected damage) approach to 'valuing the environment as input', with an application to a mangrove ecosystem in Thailand. I shall concentrate my discussion only on the production function approach, but the main methodological points raised are naturally extended to the expected damage function approach.

An environmental good or service essentially serves as a factor input into production that yields utility. The fundamental problem of the empirical application is the evaluation of the following value function:

$$V_t = E_t \sum_{j=0}^{T} \frac{B_{t+j}}{(1+r)^j}$$

where B is the social benefits in any time period of the mangrove ecosystem, and r is the discount rate.

There are two fundamental questions:

1. What is the appropriate discount rate?
2. How to evaluate B?

The first question is not explicitly addressed and different scenarios on the discount rate are adopted; the production function approach is the answer to the second question.

In theory the production function approach can be described as follows:

- Specification of a dynamic intertemporal optimization problem, where one of the constraints is the production function relating the input of interest to the measurable output.
- Solution of the model.
- Identification and estimation (or, whenever estimation is not possible, calibration) of the technology and preference parameters of the model and of the auxiliary parameters.
- Dynamic stochastic simulation of the model to derive E_tB_{t+j} and the associated confidence intervals.

In practice two alternative approaches are considered: a static one and a dynamic one. I shall not comment on the static approach because I find this inappropriate to the very nature of the problem at hand, which is, by definition, dynamic.

In the dynamic approach a model is postulated to determine the dynamics of the stock of fish measured in biomass units, X_t, the fishing effort, E_t, the landed fish price per unit harvested, p, and the harvest, h. The adopted model is described as follows:

$$X_{t+1} - X_t = F(X_t, S_t) - h(X_t, E_t)$$

$$E_{t+1} - E_t = \phi[p_t h_t - w_t E_t]$$

$$F(X_t, S_t) = rX_t\left[1 - \frac{X_t}{\alpha \ln S_t}\right]$$

$$h_t = qX_tE_t$$

$$p_t = kh_t^{\eta}$$

where $F(X_t, S_t)$ is biological growth in the current period, which is a function of S_t, the mangrove area, $h(X_t, E_t)$, harvesting is a function of the stock as well as fishing effort, E_t. Fishing effort is modelled as a partial adjustment model in which the equilibrium value is determined by fish price per unit harvested and the unit cost of effort, w.

The model is estimated and then simulated keeping w exogenous and taking alternative scenarios for S, that by consequence is taken as exogenous. Using some assumption for the discount rate the present value of a reduction in the mangrove area is then computed.

The results are interesting but there are a number of important questions that the modelling strategy leaves open:

- In the dynamic model S is exogenous and no law of motion for S is specified. The model is not capable of explaining the reduction in S that we observe in the data. In fact, if agents were acting following this model we would have never observed a reduction in S, because a reduction in S has only costs and no benefits. Macroeconometricians might see the applicability of the Lucas critique to this model as an immediate consequence of the assumption of exogeneity of S.
- Expectations do not explicitly enter the model.
- What are the costs incurred in omitting from the model the dynamics of w?
- What is the performance of this model when evaluated in sample by dynamic simulation?
- How is uncertainty added for estimation and more importantly for dynamic simulation? The result reported in Table 4 seems to take account of only coefficient uncertainty while, given the modelling choices, the fluctuations in the relevant variables not explained by the adopted model are likely to be the main source of uncertainty.

I think that the answer to this set of open questions could further enhance the potential of the interesting methods for valuing ecosystem services very well discussed in this paper.

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Edward Barbier demonstrates how basic micro theory can be implemented to estimate the value of ecological services for human welfare. In particular, two methods are developed: the production function approach and the expected damage function approach. Both methods utilize exogenous variation in the size of the ecosystem service, such as the size of a mangrove forest, on a beneficial outcome. According to the former the outcome is welfare gained from the decline in the price of a consumption good, such as fish, that utilizes the ecosystem as an input in its production process. In the latter it is the economic value of the reduction in damage, arising, for instance, from storms, that is reduced by a larger ecosystem.

The theory is rather straightforward. It is the availability of the data that the estimation depends on, and it is not clear that for most practical problems there exists sufficient exogenous variation in the ecosystem, allowing for a reliable assessment. Nevertheless, Barbier convincingly illustrates that despite the difficulties, these methods have the potential to provide important information about the value of the ecosystem and thereby the value of preventing its disappearance. This information can become critical for policymakers, and might, even if only in marginal cases, generate the crucial political force to reverse processes of natural habitat destruction.

The value of the functions of the ecosystem include, as stated by Barbier, 'climate stability, maintenance of biodiversity and beneficial species, erosion control, flood mitigation, storm protection, groundwater recharge and pollution control'. This statement reveals another limitation of the estimation methods. It focuses on a limited set of benefits, implying a potentially huge underestimation of the value of the ecosystem. First, due to information problems regarding most functions, it is difficult to identify the size of the impact and/or its welfare value. For instance, most likely exogenous variety in the ecosystem is not sufficient to estimate its effect on climate stability, and the welfare value of biodiversity is a question hard to answer.

Second, the cost of preserving the ecosystem – giving up the benefits of its alternative use – is paid by the local population, while many of its services extend beyond that. As is well known, preserving natural ecosystems is a problem with large externalities that go beyond borders. In other words, who cares? Do we expect the poor fisherman in Thailand, or their government, to allocate a significant weight in its welfare function to biodiversity? In fact, it is the population of the developed world that cares, and this population's willingness to pay a compensating price, could be above and beyond the benefit of the ecosystem to the local population.

A more technical comment on the estimation process regards the open access assumption and, in particular, the implicit assumption that changes in the habitat are sufficiently small, relative to the economy, such that the producer's surplus is unchanged. Welfare gains from a larger ecosystem emerge only from the reduction in consumer goods prices. This assumption adds to the bias in the estimation, reducing the value of the ecosystem. To see this point, suppose that prices of the consumption good are also given (traded good in an open economy). In this case there is zero welfare gain from preserving the environment.

A final comment about the estimation method regards the implicit assumption of stability of the steady-state equilibrium. However, non-monotonic convergence to the steady state might characterize the dynamics of the ecosystem. For instance, the population of a species might converge to its steady state in oscillations, implying that a negative shock to the ecosystem might, once it is sufficiently fragile, result in extinction of a species rather than a proportional reduction in the size of the natural population.

Beyond the problems of estimating the direct value of the ecological services for human welfare, lies a somewhat deeper question regarding the long-run effects of the utilization of natural resources for the benefit of mankind in the production process. Maintaining natural habitats and benefiting their production services, or destroying them and benefiting from their alternative land use for agriculture, might have different long-run consequences on demographic variables, institutional development, and, in particular, human capital promoting institutions (e.g., public schools, loans, and child labour regulation), and the resulting accumulation of human capital.

Natural resources, according to many studies, are a hurdle for the process of development, in particular the accumulation of human capital. (e.g. Gylfason, 2001). But to the best of my knowledge, we do not know yet how to make a distinction in that regard between an open-access preserved ecosystem and agricultural land. Therefore, depleting resources or increasing the size of agricultural land on the account of the ecosystem, could have a significant impact on the economy.

Moreover, the transition from an open access ecosystem into private owned farmland might have an impact on wealth inequality, in particular inequality in the ownership of such land. Deninger and Squire (1998) show that inequality in land ownership has a negative impact on economic growth. Engerman and Sokoloff (2000) provide evidence that wealth inequality, brought about oppressive institutions (e.g., restricted access to the democratic process and to education). They argue that these institutions were designed to maintain the political power of the elite and to preserve the existing inequality. Galor *et al.* (2005) provide evidence that inequality in the ownership of agricultural land has a negative effect on public expenditure on education, and argue that the elite of landowners might prevent public schooling, despite the support of the owners of capital and the working class.

On the other hand, if the destruction of the ecosystem increases farmland and thereby possibly promoting industrial development, and if the process does not generate large wealth inequality, the return to human capital will most likely rise. This could trigger a process of development stemming from reduced fertility and increased investment in education. This brings us back to the main problem of preventing the distractions of an ecosystem: the externality. Each small economy might be better-off destroying the ecosystem, giving rise to an inefficient equilibrium. The analysis suggested by Barbier, could, at least, highlight the benefits of preserving natural habitats for the local economy.

APPENDIX: APPLICATION TO THAILAND CASE STUDY

This appendix outlines the econometric estimations for valuing habitat-fishery linkages and the storm protection service of mangroves in the Thailand case study of Section 4.

A1. Static valuation of habitat-fishery linkage

To apply the static analysis of habitat-fishery linkages of Section 3.3.1 to the Thailand case study, it is necessary to estimate the unknown parameters (A, a, b) of the log-linear version of the Cobb–Douglas production function:

$$\ln h_{it} = A_0 + a \ln E_{it} + b \ln M_{it} + \mu_{it} \qquad \text{(A1)}$$

where $i = 1, \ldots, 5$ zones, $t = 1, \ldots, 14$ years (1983–96) and $A_0 = \ln A$.

Equation (A1) was estimated using the pooled data on demersal fisheries, shellfish and mangrove area from Barbier (2003). These were the data on harvest, h_{it}, and effort, E_{it}, for Thailand's shellfish and demersal fisheries, as well as mangrove area, M_{it} across the five coastal zones of Thailand and over the years1983–96. Various regression procedures for a pooled data set were utilized and compared, including: (i) ordinary least squares (OLS); (ii) one- and two-way panel analysis of fixed and random effects; and (iii) a maximum likelihood estimation by an iterated generalized least squares (GLS) procedure for a pooled time series and cross-sectional regression, which allows for correction of any groupwise heteroscedasticity, cross-group correlation and common or within-group autocorrelation. Table A1 indicates the best regression model for the shellfish and demersal fisheries respectively, and the relevant test statistics.

For demersal fisheries, the preferred model shown in Table A1 is the GLS estimation allowing for groupwise heteroscedasticity and correcting for both cross-group and common autocorrelation. For the panel analysis of the demersal fisheries, the likelihood ratio tests of the null hypothesis of zero individual and time effects across all five zones and fourteen time periods were significant, thus rejecting the null hypothesis. In addition, the Breusch–Pagan Lagrange multiplier (LM) statistic was also significant at the 95% confidence level for both the one-way and two-models, which suggests rejection of the null hypothesis of zero random disturbances. The Hausman test statistic was also significant at the 99% confidence level, suggesting that the fixed effects specification is preferred to the random effects. However, in both the one- and two-way fixed effects model the *t*-test on the estimated parameter for a in Equation (A1) was insignificant, suggesting the null hypothesis that $a = 0$ cannot be rejected. As indicated in Table A1, from the pooled time series cross-sectional GLS regression for demersal fisheries, the likelihood ratio (LR) test statistic of the null hypothesis for homoscedasticity based on the least squares regression was computed to be 24.64, which is statistically significant. Although not shown in the table, the alternative Wald test for homoscedasticity is also statistically significant and confirms rejection of the null hypothesis. Thus the GLS model with correction of groupwise heteroscedasticity is preferred to the OLS regression. The LM statistic of 14.43 also reported in Table A1 for demersal fisheries is a test of the null hypothesis of zero cross-sectional correlation, which proves to be statistically significant. Although not indicated in the table, the LR test statistic for groupwise heteroscedasticity as a restriction on cross-group correlation was estimated to be 23.26, which is also statistically significant. Thus the null hypothesis of zero cross-group correlation in the demersal fisheries regression can be rejected. The common autocorrelation coefficient across all five zones was estimated to be 0.484, and as shown in Table A1, once the GLS model for demersal fish was corrected for this common autocorrelation, the null hypothesis that the coefficient $a = 0$ is now rejected.

For shellfish, as indicated in Table A1 the preferred estimation of Equation (A1) is the GLS estimation allowing for groupwise heteroscedasticity and correcting for cross-group correlation, with A_0 restricted to zero. For the panel analysis of shellfish, the likelihood ratio tests and Breusch–Pagan LM tests of the null hypothesis of no individual and time effects were significant, thus rejecting the null hypothesis. The

Table A1. Estimates of Equation (A1) for Thailand's shellfish and demersal fisheries

Coefficient	Demersal fishery[a]	Shellfish fishery[b]
A_0	11.213 (24.568)**	–
A	0.341 (4.992)**	1.688 (38.254)**
B	0.100 (2.763)**	0.196 (3.693)**
Log-likelihood[c]	5.401	–71.517
Likelihood ratio statistic[d]	24.643**	35.076**
Lagrange multiplier statistic[e]	14.426*	21.304**

Notes: t-statistics are shown in parentheses.
[a] Preferred model is groupwise heteroscedastic and correlated GLS, corrected for common autocorrelation.
[b] Preferred model is groupwise heteroscedastic and correlated GLS, with A_0 restricted to zero.
[c] In the demersal fishery regression, correction of cross-group correlation $\mathrm{Cov}[e_{it}, e_{jt}] = \sigma_{ij}$ leads to a positive log-likelihood.
[d] Tests the null hypothesis of homoscedasticity based on OLS.
[e] Tests the null hypothesis of zero cross-group correlation based on OLS.
* Significant at 95% confidence level.
** Significant at 99% confidence level.
Sources: Author's estimations.

Hausman test statistic was significant, suggesting that the fixed effects specification is preferred to the random effects. However, in both the one- and two-way fixed effects model the t-test on the estimated parameters for a and b in Equation (A1) was insignificant, suggesting the null hypothesis $a = b = 0$ cannot be rejected. As indicated in Table A1, from the pooled time series cross-sectional GLS regression of shellfish, the LR test statistic of the null hypothesis for homoscedasticity based on the least squares regression is 35.08, which is statistically significant. Although not shown in the table, the alternative Wald test for homoscedasticity is also statistically significant and confirms rejection of the null hypothesis. Thus the GLS model with correction of groupwise heteroscedasticity is preferred to the OLS regression. The LM statistic of 21.30 also reported in Table A1 for the shellfish regression is a test of the null hypothesis of zero cross-sectional correlation, which proves to be statistically significant. Although not indicated in the table, the LR test statistic for groupwise heteroscedasticity as a restriction on cross-group correlation was estimated to be 43.90, which is also statistically significant. Thus the null hypothesis of zero cross-group correlation in the shellfish regression can be rejected. As shown in Table A1, once the GLS model of shellfish was corrected for groupwise heteroscedasticity and correlation, the null hypotheses that $a = 0$ and $b = 0$ are now rejected.

The estimations of Equation (A1) for Thailand's shellfish and demersal fisheries were used in conditions (5) and (6) to calculate the welfare impacts of mangrove deforestation over 1996–2004 on Thailand's artisanal fisheries. The analysis uses the same iso-elastic demand function as in Barbier (2003), with a demand elasticity, ε, of -0.5. The results are reported in Table 3, which shows welfare calculations for both the point estimates and upper and lower bounds on these estimates based on the standard errors of the regression coefficients reported in Table A1.

A2. Dynamic valuation of habitat-fishery linkage

The dynamic habitat-fishery modelling approach to valuing the habitat-fishery linkage is outlined in Section 3.3.2. The main difficulty in applying this approach to valuing mangrove-fishery linkages in Thailand is that data do not exist for the biomass stock, X_t, of near-shore fisheries. Thus the appropriate system of equations to estimate comprises (10) and (11). Because E_t and c_t are predetermined, both of these equations can be estimated independently (Homans and Wilen, 1997). For both the shellfish and demersal fisheries, the estimated equations are:

$$E_{it} = a_0 + a_1 R_{it-1} + a_2 E_{it-1} + \mu_{it-1} \tag{A2}$$

$$\frac{c_{it} - c_{it-1}}{c_{it-1}} = b_0 + b_1 \frac{c_{it-1}}{\ln M_{it-1}} + b_2 E_{it-1} + \mu_{it-1}, \tag{A3}$$

where $i = 1, \ldots, 5$ zones, $t - 1 = 1, \ldots, 13$ years (1983–95), $R_{it-1} = kh_{it-1}^{1+\eta}$, $a_1 = \phi$, $a_2 = (1 - \phi w)$, $b_0 = r$, $b_1 = -r/\alpha q$ and $b_2 = -q$. Both equations were estimated using the pooled data on demersal fisheries, shellfish and mangrove area from Barbier (2003). These were the data on harvest, h_{it-1}, and effort, E_{it-1}, for Thailand's shellfish and demersal fisheries, as well as mangrove area, M_{it-1} across the five coastal zones of Thailand and over the years 1983–96. In addition, to calculate R_{it-1} from h_{it-1} the elasticity $\eta = 1/\varepsilon = -2$ was assumed as in the static analysis. Various regression procedures for a pooled data set were utilized and compared, including: (i) OLS; (ii) one- and two-way panel analysis of fixed and random effects; and (iii) a maximum likelihood estimation by an iterated GLS procedure for a pooled time series and cross-sectional regression, which allows for correction of any groupwise heteroscedasticity, cross-group correlation and common or within-group autocorrelation. Tables A2 and A3 indicate the best regression models of Equations (A2) and (A3) for the shellfish and demersal fisheries respectively, and the relevant test statistics.

For demersal fisheries, the preferred model for the effort equation (A2) is the GLS estimation allowing for groupwise heteroscedasticity and corrected for common autocorrelation. For the panel analysis, the likelihood ratio and Breusch–Pagan LM tests of the null hypothesis of no individual and time effects were not significant; thus, the null hypothesis cannot be rejected. However, as indicated in Table A2, from the pooled time series cross-sectional regression of Equation (A2) for demersal fisheries, the LR test statistic of the null hypothesis of homoscedasticity based on the OLS regression is computed to be 93.22, which is statistically significant. Although not shown in the table, the alternative Wald test for homoscedasticity is also statistically significant and confirms rejection of the null hypothesis. Thus the GLS model with correction of groupwise heteroscedasticity is preferred to the OLS regression. The test statistics for the null hypothesis of zero cross-group correlation are mixed. The LM statistic of 12.24 indicated in Table A2 is significant, whereas the LR test statistic of 12.56 is not. When the GLS regression is corrected for groupwise correlation, however, the constant term a_0 is no longer significant. The common autocorrelation coefficient across all five zones is estimated to be 0.242, and although slight, correction of this autocorrelation improved the overall robustness of the GLS estimation.

As shown in Table A2, the preferred model for the effort equation (A2) for shellfish is the one-way random effects estimation corrected for heteroscedasticity. The LR and Wald tests of the pooled time series cross-sectional regressions of Equation (A2) for shellfish indicated that the null hypothesis of homoscedasticity can be rejected. Thus the GLS model with correction of groupwise heteroscedasticity is preferred to the OLS regression. However, in all versions of the GLS regression the coefficient a_1 was negative and statistically insignificant. The LR test for the presence of individual effects is statistically significant, thus rejecting the null hypothesis of no such effects, and although not shown, the equivalent *F*-test of the null hypothesis is also statistically significant. Neither the Breusch–Pagan LM test of the null hypothesis of random provincial-level disturbances nor the Hausman test of the random versus the fixed effects specification is statistically significant. Although these results are somewhat contradictory, they suggest that, if individual effects are present, they are likely to be random. The LR test and *F*-test of the presence of time effects is not significant, suggesting that the one-way is preferred to the two-way specification. Correction of heteroscedasticity improves the robustness of the one-way random effects estimation

Table A2. Estimates of Equation (A2) for Thailand's shellfish and demersal fisheries

Coefficient	Demersal fishery[a]	Shellfish fishery[b]
a_0	22.365 (2.254)*	808.720 (2.661)**
a_1	0.00004 (4.375)**	0.000003 (0.233)
a_2	0.84855 (21.703)**	0.70470 (8.183)**
Log-likelihood	–380.903	–520.513
Likelihood ratio statistic for homoscedasticity[c]	93.223**	–
Likelihood ratio statistic for correlation[d]	12.552	–
Lagrange multiplier statistic[e]	12.241*	–
Likelihood ratio statistic for individual effects[f]	–	16.285**
Breusch–Pagan Lagrange multiplier statistic[g]	–	0.04
Hausman test statistic[h]	–	1.88

Notes: *t*-statistics are shown in parentheses.
[a] Preferred model is groupwise heteroscedastic GLS, corrected for common autocorrelation.
[b] Preferred model is one-way random effects corrected for heteroscedasticity.
[c] Tests the null hypothesis of homoscedasticity based on OLS.
[d] Tests the null hypothesis of zero cross-group correlation based on OLS.
[e] Tests the null hypothesis of zero cross-group correlation based on OLS.
[f] Tests the null hypothesis of zero individual effects.
[g] Tests the null hypothesis of zero random disturbances based on OLS.
[h] Tests the null hypothesis of correlation between the individual effects and the error (i.e. random effects is preferred to fixed effects estimation).
* Significant at 95% confidence level.
** Significant at 99% confidence level.

Sources: Author's estimations.

without affecting the parameter estimates. Although not shown in the table, the preferred model displayed a very low estimated autocorrelation of 0.022.

As indicated in Table A3, the preferred model for the growth in catch per unit effort equation (A3) for demersal fisheries is the GLS estimation allowing for groupwise heteroscedasticity. For the panel analysis, the LR and Breusch–Pagan LM tests of the null hypothesis of no individual and time effects were not significant; thus, the null hypothesis cannot be rejected. However, from the pooled time series cross-sectional regression, both the LR and Wald test statistics of the null hypothesis of homoscedasticity are also statistically significant. Thus the GLS model with correction of groupwise heteroscedasticity is preferred to the OLS regression. The test statistics for the null hypothesis of zero cross-group correlation are mixed. The LM statistic of 11.03 indicated in Table A3 is significant, whereas the LR test statistic of 18.56 is not. However, correcting the GLS regression for groupwise correlation does not affect the estimation significantly. Although not shown in the table, the preferred model displayed a very low estimated autocorrelation of −0.006.

Table A3 displays the preferred model for the growth in CPE equation (A3) for shellfish, which is the GLS estimation allowing for groupwise and correlated heteroscedasticity and corrected for common autocorrelation. For the panel analysis, the LR and Breusch–Pagan LM tests of the null hypothesis of no individual and time effects were not significant; thus, the null hypothesis cannot be rejected. However, from the pooled time series cross-sectional regression, both the LR and Wald test statistics of the null hypothesis of homoscedasticity are also statistically significant. Thus the GLS model with correction of groupwise heteroscedasticity is preferred to the OLS regression. Although the LR and LM test statistics for the null hypothesis

Table A3. Estimates of Equation (A3) for Thailand's shellfish and demersal fisheries

Coefficient	Demersal fishery[a]	Shellfish fishery[b]
b_0	0.4896 (2.908)**	0.2997 (2.371)*
b_1	–0.000187 (–2.368)*	–0.000201 (–2.354)*
b_2	–0.000204 (–2.637)**	–0.000060 (–2.007)*
Log-likelihood	–22.337	–30.350
Likelihood ratio statistic for homoscedasticity[c]	24.627**	109.342**
Likelihood ratio statistic for correlation[d]	18.235	11.434
Lagrange multiplier statistic[e]	11.026*	8.491

Notes: t-statistics are shown in parentheses.
[a] Preferred model is groupwise heteroscedastic GLS.
[b] Preferred model is groupwise heteroscedastic and correlated GLS, corrected for common autocorrelation.
[c] Tests the null hypothesis of homoscedasticity based on OLS.
[d] Tests the null hypothesis of zero cross-group correlation based on OLS.
[e] Tests the null hypothesis of zero cross-group correlation based on OLS.
* Significant at 95% confidence level.
** Significant at 99% confidence level.

Sources: Author's estimations.

of zero cross-group correlation are not significant, correcting the GLS regression for groupwise correlation improves the significance confidence level of the estimated parameter b_2 from 90 to 95%. The common autocorrelation coefficient across all five zones is estimated to be 0.147, and although slight, correction of this autocorrelation improved the overall robustness of the GLS estimation.

Using the estimated parameters for Equations (A2) and (A3) for Thailand's shellfish and demersal fisheries allows simulation of the welfare impacts of mangrove deforestation over 1996–2004 on Thailand's artisanal fisheries. Again, the same demand function with elasticity of −0.5 as in Barbier (2003) is employed. The results are reported in Table 3, which shows welfare calculations for both the point estimates and upper and lower bounds on these estimates based on the standard errors of the regression coefficients reported in Tables A2 and A3.

A3. Expected damage function valuation of storm protection service

As discussed in Section 3.5, the key step in applying the expected damage function approach to valuing the storm protection service of a coastal wetland such as mangroves is to estimate how a change in mangrove area influences the expected incidence of economically damaging natural disaster events.

Suppose that for a number of coastal regions, $i = 1, \ldots, N$, and over a given period of time, $t = 1, \ldots, T$ the ith coastal region could experience in any period t any number of $z_{it} = 0, 1, 2, 3 \ldots$ economically damaging storm event incidents. A common assumption in count data models is that the count variable z_{it} has a Poisson distribution, in which case the expected number of storm events in each region per period is given by:

$$E[z_{it}|s_{it}, x_{it}] = \lambda_{it} = e^{\alpha_i + \beta_S S_{it} + \beta' x_{it}}, \quad \frac{\partial E[z_{it}|S_{it}, x_{it}]}{\partial S_{it}} = \lambda_{it}\beta_S \tag{A4}$$

where as before, S_{it} is the area of wetlands, x_{it}, are other factors, and α_i accounts for other possible 'unobserved' effects on the incidence of disasters specific to each coastal region. Estimation of β_S, along with an estimate of the conditional mean λ_{it}, allows $\partial Z/\partial S$ in Equation (13) to be determined. One drawback of the Poisson distribution (Equation (A4)) is that it automatically implies 'equidispersion', that is, the conditional variance of z_{it} is also equal to λ_{it}. To test whether this is the case, the Poisson method of estimating (A4) should be compared to other techniques, such as the Negative Binomial model, which do not assume equidispersion in the variance of z_{it}.

For the Thailand case study, the estimation of (A4) is:

$$\ln E[z_{it}|M_{it}, x_{it}] = \ln \lambda_{it} = \alpha_i + \beta_S M_{it} + \beta' x_{it} + \mu_{it}, \tag{A5}$$

where $i = 1, \ldots, 21$ coastal provinces, $t = 1, \ldots, 18$ years (1979–96). The EM-DAT (2005) International Disaster Database contains data on the number of coastal disasters occurring in Thailand since 1975 and the approximate location and date of its impacts. From these data it is possible to determine z_{it}, the number of economically damaging coastal natural disasters that occurred per province per year over 1979–96. Mangrove area, M_{it}, is measured in terms of the annual mangrove area in square kilometres for each of the 21 coastal provinces of Thailand over 1979–96. Two control variables were included as the additional factors, x_{it}, which may explain the incidence of economically damaging coastal disasters, the population density of a province and a yearly time trend variable. The inclusion of the population density variable reflects the prevailing view in the natural disaster management literature that 'hazard events that occur in unpopulated areas and are not associated with losses do not constitute disasters' (Dilley *et al.*, 2005, p. 115).[11] The yearly time trend was included as a control because the number of coastal natural disasters seems to have increased over time in Thailand (see Figure 5).[12]

Various regression procedures for a panel data set for count data models were utilized and compared, including: (1) Poisson models assuming equidispersion, i.e. equality of the conditional mean and the variance; (2) maximum likelihood estimation of Negative Binomial models allowing for unequal dispersion; and (3) comparing provincial to zonal fixed effects. Table A4 reports the best count data model for estimating Equation (A5) and the relevant test statistics.

As shown in Table A4, the preferred specification of the count data model is the Negative Binomial model with zonal fixed effects. In both the Poisson and Negative Binomial panel models, the zonal fixed effects specification (with coastal zone 5 as the default) is preferred to individual province effects, which is verified by LR tests of the two specifications. Although the parameter estimates for the zonal fixed effects are not shown, these estimated effects were significant at the 95% or 99% confidence levels. As indicated in Table A4, two standard tests were employed for the null hypothesis of equidispersion of the conditional mean and variance of the Poisson specification of the count data model (Cameron and Trivedi, 1998; Greene, 2003). Both the LM statistic and the *t*-test for equidispersion based on the residuals of the Poisson regression are significant, indicating that the null hypothesis can be rejected, and the Negative Binomial model that does not assume equidispersion is preferred to the Poisson specification. The LR statistic reported in the table tests the null hypothesis that the coefficients of the regressors are zero; as the statistic is significant, the hypothesis is rejected.

The results displayed in Table A4 for the preferred model show that a change in mangrove area has a significant influence on the incidence of coastal natural disasters in Thailand, and with the predicted sign. The point estimate for β_S indicates that a 1 km^2 decline in mangrove area increases the expected number of disasters by 0.36%.

[11] This view is also reflected in the criteria used in the International Disaster Database to decide which hazard events should be recorded as 'natural disasters'. In order for EM-DAT (2005) to record an event as a disaster, at least one or more of the following criteria must be fulfilled: 10 or more people reported killed; 100 people reported affected; declaration of a state of emergency; call for international assistance. The simple correlation between population density and mangrove area for the sample is relatively low (−0.389).

[12] This is a procedure recommended by Rose (1990), when such a trend effect is suspected.

Table A4. Negative binomial estimation of Equation (A5) with zonal fixed effects

Variable	Parameter estimate[a]	Marginal effect[b]
Mangrove area (M_{it})	−0.0036 (−4.448)**	−0.0031 (−2.745)**
Population density ($POPDEN_{it}$)	−0.0005 (−1.079)	−0.0004 (−0.894)
Annual time trend ($YRTRN_{it}$)	0.0781 (5.558)**	0.0669 (2.615)**
Dispersion parameter (α_{it})	0.0001	
Estimated conditional mean (λ)	0.8559	
Log-likelihood	−373.66	
Lagrange multiplier statistic[c]	39.967**	
Regression t-test[d]	−5.385**	
Likelihood ratio statistic[e]	74.919**	

Notes: t-statistics shown in parentheses.
[a] Parameter estimates for the zonal fixed effects are not shown. Zone 5 is the default and the fixed effects for zones 1 to 4 were negative and significant at the 95% or 99% confidence levels.
[b] Estimate of $\lambda_{it}\beta_3$ (see Equation (A4)).
[c] Tests the null hypothesis of equidispersion in the Poisson model.
[d] A regression-based test of the null hypothesis of equidispersion in the Poisson model.
[e] Tests the null hypothesis that the restricted regression without the explanatory variables M_{it}, $POPDEN_{it}$ and $YRTRN_{it}$ is the preferred Negative Binomial model with zonal fixed effects.
* Significant at 95% confidence level.
** Significant at 99% confidence level.
Sources: Author's estimations.

It is likely that the mangrove loss in Thailand, especially since the mid-1970s (see Figure 4), has increased the expected number of economically damaging coastal natural disasters per year. The estimated marginal effect corresponding to β_S of a change in mangrove area on coastal natural disasters (–0.0031) can be employed to estimate the resulting impact of mangrove deforestation over 1979–96 in Thailand on expected damages of natural coastal disasters. This is described further in Section 4.3 and shown in Table 4.

As discussed in Section 3.5, an underlying hypothesis of the expected damage function methodology is that, if coastal wetland loss increases the incidence of natural disaster per year, then wetland loss is also associated with increasing storm damages. However, under certain circumstances the results of a count data model could provide a misleading test of this null hypothesis. For instance, suppose a loss in wetland area is associated with a change in the incidence of storms from one devastating storm to two relatively minor storms per year. The count data model would then be interpreted as not providing evidence against the null that the change in the wetland area increases expected storm damages, when what has actually happened is that total storm damages have declined over time with wetland loss. This suggests the need for a robustness check on the count data model, such as Equation (A5) in the Thailand case study, to ensure that such situations do not dominate the application of the EDF approach.

One possible robustness check is to test the null hypothesis directly; that is, are total damages from storm events increasing with coastal wetland loss? In the Thailand case study, the relevant estimation is

$$D_{it} = \alpha_i + \beta_S M_{it} + \beta' x_{it} + \mu_{it} \tag{A6}$$

where the dependent variable, D_{it}, is total real damages from all storm events per province per year over 1979–96. The EM-DAT (2005) database provides data on the total economic damages per province per year in Thailand, and these data were deflated using the 1996 GDP deflator. The standard regression procedures for the panel analysis of Equation (A6) were performed, including comparing OLS with fixed and random effects. Table A5 reports the OLS and random effects specifications for the preferred version of Equation (A6).

The preferred model in Table A5 is the pooled weighted least squares estimation with correction for heteroscedasticity. The LR test of the null hypothesis of zero individual effects across all 21 provinces is not statistically significant. Although not shown in the table, an alternative *F*-test of the null hypothesis is also not significant. Neither the Breusch–Pagan LM test of the null hypothesis of random provincial-level disturbances nor the Hausman test of the random versus the fixed effects specification is statistically significant. These tests confirm that in the panel analysis of Equation (A5) of the weighted OLS regression is more efficient than either the random or fixed effects models.

The weighted least squares regression in Table A5 indicates that, over 1979–96 and across the 21 coastal provinces of Thailand, total real storm damages increased with mangrove loss. The point estimate suggests that a 1 km^2 decline in mangrove area increases real storm damages by around \$52 per province per year. The regression also confirms that, for the Thailand case study, the null hypothesis that storm damages increase with mangrove loss cannot be rejected.

Table A5. Panel estimation of Equation (A6) for total storm damages, Thailand

Variable[a]	Pooled OLS[b]	Random effects[b]
Mangrove area (M_{it})	−51.527 (−1.976)*	−52.378 (−1.563)
Population density ($POPDEN_{it}$)	−12.896 (−0.343)	−18.723 (−0.395)
Annual time trend ($YRTRN_{it}$)	965.325 (2.058)*	983.653 (2.100)*
Constant	1 3748.820 (1.275)	1 4728.707 (1.153)
Log-likelihood	−4598.425	
Likelihood ratio statistic[c]	14.173	
Lagrange multiplier statistic[d]		2.24
Hausman test[e]		1.04

Notes: *t*-statistics shown in parentheses.

[a] Parameter estimates for the zonal fixed effects for Zone 1 and Zone 4 are not shown. Although neither parameter was statistically significant, their inclusion improved the robustness of the overall regression.

[b] Weighted least squares with robust covariance matrix to correct for heteroscedasticity.

[c] Tests the null hypothesis of no fixed provincial effects.

[d] Tests the null hypothesis of no random provincial effects.

[e] Tests the null hypothesis that the random effects specification is preferred to the fixed effects. Test was performed excluding the zonal fixed effect for Zone 4.

* Significant at 95% confidence level.

Sources: Author's estimations.

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