Environmental Project Evaluation in Developing Countries:

Valuing the Environment as Input

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Non-Technical Summary

The following paper suggests that an important evaluation problem in developing countries is assessing the value and optimal use of environmental resources as inputs into production. Essentially, this amounts to determining the best use of land, water and other resources. The paper outlines the simple methodology behind the general evaluation approach, e.g. ensuring the most efficient use of a resource requirese incorporating environmental costs and benefits, and comparing the returns from competing uses. However, implementing this approach raises important issues of valuing the role of ecological services and resources in contributing to, supporting and protecting economic activity. To illustrate these issues, the paper draws on several case studies, such as allocating agricultural land among competing uses (e.g. gum arabic versus annual crops in Sudan), diversion of water from downstream floodplains for upstream water developments (e.g. Hadejia-Jama'are floodplain, Northern Nigeria), and evaluating the benefits of ecological services in support of economic activity (e.g. mangrove-fishery linkages in Campeche, Mexico).

The gum arabic case study from Sudan is a straightforward application of the production function approach. The *Acacia senegal* tree is the key environmental input, which in turn yields an important marketed output, gum arabic resin. However, the *Acacia senegal* tree also produces many additional environmental benefits, and a full analysis of the economic value of farming systems incorporating this tree needs to take into account these wider benefits as well. As this case study indicates, although it is reasonably easy to compare the marketed returns of gum arabic systems to the returns of competing agricultural systems, the wider environmental benefits of gum arabic systems need also to be assessed.

The diversion of water from downstream floodplains for upstream water developments is illustrated with the example of the Hadejia-Jama'are floodplain, Northern Nigeria. A combined economic and hydrological analysis was conducted to simulate the impacts of these upstream projects on the flood extent that determines the downstream floodplain. The economic gains of the upstream water projects were then compared to the resulting economic losses to downstream agricultural, fuelwood and fishing benefits. The results confirm that in all the scenarios simulated the additional value of production from large-scale irrigation schemes does not replace the lost production attributed to the wetlands downstream. Gains in irrigation values account for at most around 17% of the losses in floodplain benefits. Although the existing upstream water development projects cannot be reversed, the analysis supports redesigning some major dams to allow regulated water releases to conserve as much of the remaining floodplain area as possible.

The production function approach can also be incorporated into intertemporal models of renewable resource harvesting in cases where the ecological function affects the growth rate of a stock over time. In such cases, the production function link is a dynamic one, as the ecological function affects the rate at which a renewable resource increases over time, which in turn affects the amount of offtake, or harvest, of the resource. The case study of mangrove-fishery linkages in Campeche, Mexico illustrates how this approach can be applied, and estimates the economic contribution of mangroves as a breeding ground and nursery for an offshore shrimp fishery. However, the open access conditions prevailing in the fishery dominate, and any increase in harvest and revenues from an expansion in mangrove area is likely to be short-lived, as it would simply draw more effort into the fishery. Better management of the Campeche shrimp fishery is therefore essential to realizing the more long-term economic benefits of protecting mangrove habitat.

Abstract

An important evaluation problem in developing countries is assessing the value and optimal use of environmental resources as inputs into production. This is a key methodology required for many general evaluation approaches, e.g. ensuring the most efficient use of a resource requirese incorporating environmental costs and benefits, and comparing the returns from competing uses. However, implementing this approach requires valuing the role of ecological services and resources in contributing to, supporting and protecting economic activity. To illustrate these issues, the paper draws on several case studies, such as allocating agricultural land among competing uses (e.g. gum arabic versus annual crops in Sudan), diversion of water from downstream floodplains for upstream water developments (e.g. Hadejia-Jama'are floodplain, Northern Nigeria), and evaluating the benefits of ecological services in support of economic activity in a dynamic context (e.g. mangrove-fishery linkages in Campeche, Mexico).

JEL classification: O1, Q0, Q2.

Keywords: developing countries, evaluation, ecological services, environmental input, production function approach, project appraisal, valuation.

Introduction

Natural resource management is crucial to the developing economies of the world. These economies, especially the lower-income countries, are highly dependent on primary production as the source of long-term, sustainable economic development (Barbier 1994). Successful exploitation of primary production - agriculture, fishing, forestry and minerals - in turn depends on efficient and sustainable management of the resource base supporting primary productive activities. Moreover, as developing countries industrialize and as their populations concentrate in urban settlements, the role of the environment in assimilating waste products and providing life-support amenities will become increasingly important. Protection and conservation of key natural systems and important ecological functions will also be essential, not only in terms of their potential value in terms of recreation and tourism but also because these systems and functions may provide invaluable support and protection for economic activity and human welfare.

Any analysis of the contribution of environmental resources to development must invariably involve valuing the key economic functions performed by these resources, and in turn, the impact of economic activity on the environment. *Valuation of environmental resources and impacts* is the important starting point for the application of environmental economics in developing countries, and is essential for comparing the economic and social returns to economic activities and projects that involve the use of environmental resources.

The following paper examines the critical valuation issue of assessing the role of ecological resources and services in supporting and protecting economic activity in developing countries. This valuation issue is important for two principal reasons. First, as the economic contribution of environmental resources and systems is often nonmarketed, it is generally ignored in investment decisions and policies that affect the allocation of these resources and systems. Second, failure to take into account such environmental values may distort development and investment decisions, leading to unnecessary and inefficient depletion, degradation and over-exploitation of natural resources and environments. The result can be a net loss of economic welfare to a developing country.

However, assessing the role of the environment as an input into economic activities in developing countries raises some interesting methodological challenges. The direct exploitation or use of a environmental resources is often fairly straightforward to value, but valuing the indirect use of ecological resources and services in terms of providing external support and protection of economic activities elsewhere can be more problematic. To illustrate the various approaches and issues involved, the following paper draws on several case studies, such as allocating land among competing uses (e.g. gum arabic versus annual crops in Sudan), diversion of water from downstream floodplains for upstream water developments (e.g. Hadejia-Jama'are floodplain, Northern Nigeria), and evaluating the benefits of ecological services in support of economic activity (e.g. mangrove-fishery linkages in Campeche, Mexico).

Comparing Returns from Competing Environmental Uses

Many large-scale investment projects and programmes, such as hydroelectric dams, irrigation schemes, commercial agricultural development schemes, road building, and so on, have significant environmental impacts. Some of these impacts may impose additional costs or benefits on society.

In recent years, advances have been made in applying economic valuation techniques to analyzing the environmental impacts of investment projects and programmes in developing countries (Barbier *et al.* 1997; Dixon *et al.* 1989).

Failure to account fully for the environmental impacts of an investment project or programme means that its net economic worth is being misrepresented. Often, when significant external environmental costs are present, the result is a misallocation of resources and excessive environmental degradation. These additional costs must be included as part of the costs of the development investment.

For example, assume that there is an upstream development project on a river that is providing water for agriculture. Given *direct benefits* (e.g., irrigation water for farming), B^D, and *direct costs* (e.g., costs of constructing the dam, irrigation, channels, etc.), C^D, then the *direct net benefits* of the project are:[1]

$$NB^D = B^D - C^D. \tag{1}$$

However, by diverting water that would otherwise flow into downstream wetlands, the development project may result in losses to floodplain agriculture and other primary production activities (e.g., fishing, fuelwood, livestock grazing), less groundwater recharge and other external impacts. Given these reductions in the *net production and environmental benefits*, NB^E, of the wetlands, then the true net benefits of the development project (NB^P) are NB^D - NB^E. The development project can therefore only be acceptable if:

$$NB^P = NB^D - NB^E > 0. \tag{2}$$

Thus, in the presence of significant environmental impacts, the net benefits of a development project or programme cannot be appraised in terms of its direct benefits and costs alone. The forgone net benefits of disruption to the natural environment and degradation must also be included as part of the opportunity costs of the development investment.

Valuing the Environment as Input

Assessment of the net production and environmental benefits, NB^E, is of course critical to implementing the above decision rule (2). However, the analysis of such benefits is not always so straightforward. Although many production and environmental benefits directly or indirectly support economic activity, there are many ways in which such use values occur.

For example, direct uses of the environment would include both *consumptive uses* of resources (e.g. livestock grazing, fuelwood collection, forestry activities, agriculture, water use, hunting and fishing) and *non-consumptive uses* of environmental 'services' (e.g. recreation, tourism, *in situ* research and education, navigation along water courses). Direct uses of natural resources and systems could involve both commercial and non-commercial activities, with some of the latter activities often being important for the subsistence needs of local populations. Commercial uses may be important for both domestic and international markets. In general, valuing the marketed products (and services) of the environment is easier than valuing non-commercial and subsistence direct uses.

When the environment is being *indirectly used*, in the sense that the ecological functions of the environment are effectively supporting or protecting economic activity, then the value of these functions are essentially nonmarketed. This value arises out of the natural 'interaction' between different ecological systems and processes; in particular, the ecological functioning of one ecosystem may affect the functioning and productivity of an adjacent system that is being exploited economically. As a result, the overall productivity and stability of the latter ecosystem may be critically dependent on the maintenance of a few key *external support functions* provided by the neighbouring ecosystem. Examples include the role of coastal marshland and mangrove systems as breeding grounds and nurseries for off shore fisheries; flood and sedimentation control provided by upper watershed and montane forest systems; sediment and nutrient retention by riverine wetlands; and semi-arid and arid brush forests protecting against desertification of rangelands. As the economic contribution to production activities of these external ecological support functions are non-marketed, go financially unrewarded and are only indirectly related to the economic activities that they protect or support, the indirect use values of external ecological support functions are often extremely difficult to value. They are also good examples of how the existence of *ecological externalities* also gives rise to *economic externalities* (Barbier 1998).

In recent years, our understanding of the direct and indirect use of the environment for economic activities and our valuation techniques for assessing these economic contributions have improved greatly. In particular, economists have demonstrated that it is possible to value nonmarketed environmental resources and services through the use of *surrogate market valuation*, which essentially uses information about a marketed good to infer the value of a related nonmarketed good (Freeman 1993). Travel cost methods, recreational demand analysis, hedonic pricing and averting behaviour models are all examples of surrogate market valuation that attempt to estimate the derived demand by households for environmental quality.

However, this paper describes another type of surrogate market valuation that is particularly useful for the valuation of nonmarketed values associated with biological resources and ecosystems that make a direct or indirect contribution to economic activity. This is often referred to in literature as either the *production function approach* or *valuing the environment as input* (Aylward and Barbier 1992; Barbier 1994; Freeman 1993; Mäler 1991).[2] This approach recognizes that ecological resources and functions may have an economic value that arises through their support of economic production and human welfare, or through protection of valuable assets and property. Moreover, given the direct dependence of many production systems and economic livelihoods in developing countries on natural resources and systems, valuing the environment as input is considered to widely applicable to many important economic development and investment decisions in those countries (Aylward and Barbier 1992; Barbier 1994; Mäler 1991).

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

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

$$Q = F(X_i...X_k, S), \qquad \textbf{(3)}$$

For example, suppose that the ecological function of particular interest is the role of mangroves in supporting off-shore fisheries through serving both as a spawning ground and a nursery for fry. The area of mangroves in a coastal region, S, may therefore have a direct influence on the catch of mangrove-dependent species, Q, which is independent from the standard inputs of a commercial fishery, $X_i...X_k$. Including mangrove area as a determinant of fish catch may therefore 'capture' some element of the economic contribution of this important ecological support function. That is, the if the impacts of the change mangrove area 'input' can be estimated, it may be possible to indicate how these impacts influence the marginal costs of production. As shown in Figure 1, for example, an increase in wetland area increases the abundance of crabs and thus lowers the cost of catch. The value of the wetlands' support for the fishery - which in this case is equivalent to the value of increments to wetland area - can then be imputed from the resulting changes in consumer and producer surplus.

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

In addition, as pointed out by Freeman (1991), market conditions and regulatory policies for the marketed output will influence the values imputed to the environmental input. For example, in the previous example of mangroves supporting an off shore fishery, the fishery may be subject to open access conditions. Under these coditions, rents in the fishery would be dissipated, and price would be equated to average and not marginal costs. As a consequence, producer surplus is zero and only consumer surplus determines the value of increased wetland area (see Figure 1).

Applications of the production function approach may be most straightforward in the case of *single use systems* - i.e. resource systems in which the predominant economic value is a single regulatory function, or a group of ecological functions providing support or protection for an economic activity in concert. In the case of *multiple use systems* - i.e. resource systems in which a regulatory function may support or protect many different economic activities, or which may have more than one regulatory ecological function of important economic value - applications of the production function approach may be slightly more problematic. In particular, assumptions concerning the ecological relationships among these various multiple uses must be carefully constructed to avoid problems of *double counting* and *trade offs* between the different values (Aylward and Barbier 1992).

Finally, for some valuation problems, choosing whether to incorporate intertemporal aspects of environmental change can be important. For example, there are two ways in which the production function approach has been implemented to estimate the value of estuarine wetlands and mangroves in supporting off-shore fisheries, (Barbier 1997; Freeman 1993). The first is essentially a *static* approach, which either ignores the intertemporal fish harvesting process (i.e. assumes single-period or static production) or assumes that fish stocks are always constant (i.e. harvesting always offsets any natural growth in the fish population). Either assumption can be used to derive a market equilibrium for fish harvest, and thus to estimate changes in consumer and producer surplus arising from the impacts of a change in mangrove area on this static equilibrium. The second is essentially a *dynamic* approach, which attempts to model the effects of a change in mangrove area on the growth function of the intertemporal fishing problem. By solving for the long run equilibrium of the fishery, the comparative static effects and resulting welfare impacts of a change in mangrove area on the equilibrium levels of stock, effort and harvest can be determined.

The rest of the paper uses case studies from developing countries to illustrate the application of the production function approach to valuing the environment as an input into economic activitiy. Three case studies are chosen: the allocation of agricultural land between gum arabic systems versus competing annual crops in Sudan; diversion of water from downstream floodplains for upstream water developments in the Hadejia-Jama'are floodplain, Northern Nigeria; and evaluating mangrove-fishery linkages in Campeche, Mexico.

Allocating Land among Competing Uses[3]

The following case study is a straightforward application of decision rule (2) through use of the above production function approach. In this case, a particular tree species, *Acacia senegal*, is the key environmental input, which in turn yields an important marketed output, gum arabic resin. However, the *Acacia senegal* tree also produces many additional environmental benefits, and a full analysis of the economic value of farming systems incorporating this tree needs to take into account these wider benefits as well. As this case study indicates, although it is reasonably easy to compare the marketed returns of gum arabic systems to the returns of competing agricultural systems, the wider environmental benefits of gum arabic systems need also to be assessed.

Case study: gum arabic versus annual crops, Sudan

The gum arabic tree (*Acacia senegal*) is a naturally occurring species in the Sudano-Sahel region that serves a variety of valuable economic and ecological functions. The gum produced by the tree - the 'gum arabic' - is widely sought after in importing countries for use as a emulsifier in confectionery and beverages, photography, pharmaceuticals and other manufacturing industries. In addition, the *Acacia senegal* tree provides fodder for livestock, fuelwood and shade. There are also numerous indirect benefits associated with these trees; for example, its extensive lateral root system reduces soil erosion and runoff, as a leguminous tree it fixes nitrogen which encourages grassy growth for livestock grazing, it serves as a windbreak and is important in dune fixation. For these reasons, the tree is the preferred species in bush-fallow rotation and intercropping farming systems prevalent in the arid West of Sudan. On a larger scale, across the Sudano-Sahelian region, the gum arabic 'belt' acts as a buffer against desertification.

The choice by farmers in Sudan to incorporate gum arabic trees in their farming systems will depend on whether the above benefits from a gum arabic-based system exceed those of alternative systems. The potentially high financial rate of return to a gum arabic based farming system coupled with its important environmental benefits would seem to imply that such a system would be ideal for combatting desert encroachment and rehabilitating the gum belt of Sudan. However, in recent decades, distortionary government policies have meant that the real producer price of gum arabic in Sudan has fluctuated considerably, as has the relative price of gum to its competitor cash crops (sesame, groundnuts) and even food crops (sorghum, millet). Farmers' share of the export value of gum also remains low, which has been one reason for the recent increase in smuggling of gum to neighbouring countries.

An economic analysis was conducted of six representative cropping systems containing gum arabic cultivation in Sudan (Barbier 1992). The results are indicated in Table 1. Although the analysis shows that all six systems are economically profitable, the relative profitability of gum compared to other crops in each system is generally lower than that of other crops - except in the Tendelti system of the White Nile where field crop damage occurs frequently. In most systems there are initial losses due to the need to establish gum gardens before they begin producing. This would suggest that maintaining the real producer price of gum by removing distortions in the market, as was assumed in the analysis, is a necessary economic incentive to encourage gum cultivation.

The analysis in Table 1 suggests that it may be more economically profitable to replace the gum arabic component in each of the six systems with annual crops. However, despite the lower relative returns to cultivating gum arabic, there are several reasons why converting land under *A. Senegal* to cultivating annual crops may not be desirable:

- some land under *A. Senegal* may not be suitable for growing annual crops, resulting ins very low and fluctuating yields

- fallowing land may be important to maintain its fertility; gum arabic is the ideal cash crop for this purpose

- the environmental benefits of gum arabic trees (e.g. control of erosion/runoff, wind breaks, dune fixation, nitrogen fixation) were not included in the analysis, and these may be significant in maintaining the yields of field crops within the farming systems

- the role of the gum belt in controlling desertification certainly is significant in supporting and protecting farming systems in the region, and although this collective benefit cannot be captured in an analysis of individual systems, traditional farming communities in the gum belt region are very much aware of this benefit

- risk-averse farmers may desire some of their land being held under gum cultivation, because the returns to gum - although lower than the maximum expected returns from the cash crops - may be less variable under stressful environmental and climatic conditions

- gum cultivation provides cash income to farmers outside of the growing season for cash crops.

Thus, a fuller economic assessment of the wider environmental benefits of gum arabic systems, and not just the returns to marketed gum, would perhaps reveal the true economic value of these systems compared to annual crops.

Upstream Water Diversion

As discussed above, the type of ecological and economic externalities associated with upstream water diversion are fairly straightforward, 'unidirectional' problems. Water that is diverted upstream means less water downstream, and the result is that upstream activities benefit at the expense of downstream activities. Determining the optimal level of water diversion between upstream and downstream uses is therefore a critical issue. As Barbier (1998) has shown, by extending decision rule (2) into a simple production function model of the river basin, it is possible to illustrate the key conditions underlying this important allocation decision. A case study of the impacts of upstream water project developments on the downstream floodplain in the Hadejia-Jama'are River Basin in Northern Nigeria also illustrates the valuation and incentive issues associated with this particularly type of externality.

The basic river basin model developed by Barbier (1998) considers two competing uses for water flowing through the basin: water diverted upstream for irrigated agriculture and water flowing downstream for natural floodplain agriculture. Thus floodplain agricultural production is entirely dependent on the stock of water available downstream and 'stored' in the naturally occuring flooplain. This downstream water supply, W, is freely available to the agricultural system, which uses a fixed proportion of it, kW.[4] Output per hectare in the agroecosystem, h_1, is therefore a function of the available water stock, W, and a vector of other inputs, z_1 (e.g. labor, purchased inputs, etc.). Assuming that the agricultural system produces for a market in response to a given market price, p^h, and faces a vector of given input prices, c, the discounted economic returns of present and future production in the agroecosystem can be represented as

$$V^1 = V^1(z_1, W, t; p^h, c, \delta) = \int_0^T (p^h h_1 - c z_1) e^{-\delta t} dt$$
$$h_1 = h_1(z_1, W),\ h_{1i} > 0,\ h_{1ii} < 0,\ i = z_1, W, \qquad \textbf{(4)}$$

where δ is the rate of discount and T is the time period over which the downstream agroecosystem is in operation.

However, the continuous diversion of water for upstream projects, such as irrigation or water supply, reduces the flow of water through the watershed and drainage basin, directly affecting the supply of water available downstream for the floodplain. This diversion can be represented by:

$$W(t) - W(0) = -\int_0^t d\, dt$$
$$\text{or} \quad \dot{W} = -d. \qquad \textbf{(5)}$$

Thus d represents the amount of water diverted each period by upstream projects, and hence the amount of water no longer available for the stock of downstream supply, W. If it is assumed that water in a river basin is being continuously diverted upstream for an irrigated agricultural system, then the discounted returns per hectare for this system, V^2, may take the following form

$$V^2 = V^2(z_2, d, t; p^h, c, \delta) = \int_0^T (p^h h_2 - cz_2)\, e^{-\delta t}\, dt \qquad \textbf{(6)}$$

$$h_2 = h_2(z_2, d, \textstyle\int d\, dt) = h_2(z_2, d, W(0) - W(t)),\; h_{2i} > 0,\; h_{2ii} < 0,\; i = z_2, d, W(0) - W(t),$$

where agricultural yield, h_2, is presumed to be an increasing function of both current water diversion, d, cumulative diversion into the upstream irrigation network, $\int d\, dt$, and variable inputs, z_2. Output is sold at the given market price for irrigated crops, p^h, and cost of inputs is c. In this simple example, it is assumed that there is no user charge imposed on the agricultural system for the irrigation water supplied through the upstream water project.[5]

In this case, the economic externality problem arises because the farmer benefiting from the diverted irrigation from the upstream water project is unconcerned about any resulting impacts on downstream water availability. For example, it is clear from equation (6), that the economic agent would maximize discounted returns by choosing a rate of water diversion, d^i, in each period t, that would satisfy the following condition

$$\frac{\partial V^2}{\partial d} = p^h h_{2d}(d^i) = 0, \qquad \forall\ t \qquad \textbf{(7)}$$

i.e., from the standpoint of the upstream irrigation farmer, optimal diversion of water to the upstream irrigated agricultural system in every time period should occur until there are no more gains to be had from using additional water inputs.

In contrast, if there was a single river basin planning authority, this agency would be concerned with maximizing not only the returns to upstream irrigated agriculture, e.g.equation (6), but also the returns to the downstream agroecosystem as well, e.g. equations (4) and (5). Denoting λ as the costate variable, or 'shadow price', of the downstream water supply, the current value Hamiltonian, H, and relevant first order conditions of concern to the river basin authority might be:

$$
\begin{gathered}
H = (p^h h_1 - cz_1) + (p^h h_2 - cz_2) - \lambda d \\
\frac{\partial H}{\partial d} = p^h h_{2d}(d^*) - \lambda = 0 \\
-\frac{\partial H}{\partial W} = \dot{\lambda} - \delta\lambda = p^h h_{2W(0)-W(t)} - p^h h_{1W} \quad \text{or} \quad \frac{\dot{\lambda}}{\lambda} + \frac{h_{1W}}{h_{2d}} - \frac{h_{2W(0)-W(t)}}{h_{2d}} = \delta. \quad \forall t
\end{gathered}
\tag{8}
$$

In comparison with (7), the first order conditions in (8) show that the optimal rate of water diversion for the entire river basin occurs where the marginal benefit to upstream irrigated agriculture of diversion equals the shadow price, or value, of water supply to the downstream agroecosystem. By definition, $\lambda(t)$ is 'shadow price', or imputed value, of the downstream water stock, W, in terms of the net returns to the agroecosystem in the lower catchment^t. As this shadow value is positive, and is likely to be increasing over time as W is depleted, then the optimal rate of diversion, d^*, for the entire watershed and drainage basin will be less than the rate, d^i, that an upstream farmer would decide on his or her own.[6]

Consequently, the *valuation* problem facing the river basin planner is to determine the contribution of the available downstream water to the returns to the lower catchment agricultural system, and how these returns might change over time as more water is diverted to the upstream project. Again, a basic production function approach to estimate downstream floodplain benefits, as indicated by equation (3), is critical. As the following example indicates, overcoming this incentive problem and ensuring an optimal rate of diversion in a river basin system is particularly difficult if plans and development for upstream water diversion are already well advanced.

Case study: Hadejia-Jama'are River Basin, northern Nigeria[7]

In Northeast Nigeria, an extensive floodplain has been created where the Hadejia and Jama'are Rivers converge to form the Komadugu Yobe River which drains into Lake Chad. Although referred to as wetlands, much of the Hadejia-Jama'are floodplain is dry for some or all of the year. Nevertheless, the floodplain provides essential income and nutrition benefits in the form of agriculture, grazing resources, non-timber forest products, fuelwood and fishing for local populations. The wetlands also serve wider regional economic purposes, such as providing dry-season grazing for semi-nomadic pastoralists, agricultural surpluses for Kano and Borno states, groundwater recharge of the Chad Formation aquifer and 'insurance' resources in times of drought. In addition, the wetlands are a unique migratory habitat for many wildfowl and wader species from Palaearctic regions, and contain a number of forestry reserves.

However, in recent decades the Hadejia-Jama'are floodplain has come under increasing pressure from drought and upstream water developments. The maximum extent of flooding has declined from between 250,000 to 300,000 ha in 1960s and 1970s to around 70,000 to 100,000 ha more recently. Drought is a persistent, stochastic environmental problem facing all sub-Saharan arid and semi-arid zones, and the main cause of unexpected reductions in flooding in drought years. The main long-term threat to the flooplain is water diversion through large-scale water projects on the

Hadejia and Jama’are Rivers. Upstream developments are affecting incoming water, either through dams altering the timing and size of flood flows or through diverting surface or groundwater for irrigation. These developments have been taking place without consideration of their impacts on the Hadejia-Jama'are floodplain or any subsequent loss of economic benefits that are currently provided by use of the floodplain.

The largest upstream irrigation scheme at present is the Kano River Irrigation Project (KRIP). Water supplies for the project are provided by Tiga Dam, the biggest dam in the basin, which was completed in 1974. Water is also released from this dam to supply Kano City. The second major irrigation scheme within the river basin, the Hadejia Valley Project (HVP), is under construction. The HVP will be supplied by Challawa Gorge Dam on the Challawa River, upstream of Kano, which was finished in 1992. Challawa Gorge may also provide water for Kano City water supply. A number of small dams and associated irrigation schemes have also been constructed or are planned for minor tributaries of the Hadejia River. In comparison, the Jama’are River is relatively uncontrolled with only one small dam across one of its tributaries. However, plans for a major dam on the Jama’are at Kafin Zaki have been in existence for many years, which would provide water for an irrigated area totalling 84 000 ha. Work on Kafin Zaki Dam has been started and then stopped a number of times, most recently in 1994, and its future is at present unclear.

A combined economic and hydrological analysis was recently conducted to simulate the impacts of these upstream projects on the flood extent that determines the downstream floodplain area (Barbier and Thompson 1998). The economic gains of the upstream water projects were then compared to the resulting economic losses to downstream agricultural, fuelwood and fishing benefits.

Table 2 indicates the scenarios that comprise the simulation. Since Scenarios 1 and 1a reflect the conditions without any of the large-scale water resource schemes in place within the river basin they are employed as baseline conditions against which Scenarios 2-6 are compared. Scenario 2 investigates the impacts of extending the KRIP-I to its planned full extent of 22 000 ha without any downstream releases. In contrast, Scenario 3 simulates the impacts of limiting irrigation on this project to the existing 14 000 ha to allow a regulated flood from Tiga Dam in August to sustain inundation within the downstream Hadejia-Jama'are floodplain. Challawa Gorge is added in Scenario 4 and the simulated operating regime involves the year-round release of water for the downstream HVP, but not for sustaining the Hadejia-Jama'are floodplain. Scenario 5 simulates the full development of the four water resource schemes without any releases for the downstream floodplain. In direct comparison, Scenario 6 shows full upstream development, but less upstream irrigation occurs in order to allow regulated water releases from the dams to sustain inundation of the downstream floodplain.

In Table 3.a., the impacts of Scenarios 2-6 upon peak flood extent downstream are evaluated as the difference between maximum inundation predicted under each of these scenarios and the peak flood extents of the two baseline scenarios. The gains in upstream irrigated area are also indicated for each scenario in Table 3.a. The estimated floodplain losses are indicated in Table 3.b. for each scenario compared to the baseline Scenarios 1 and 1a. Given the high productivity of the floodplain, the losses in economic benefits due to changes in flood extent for all scenarios are large, ranging from US$2.6-4.2 million to US$23.4-24.0 million.[8] As expected, there is a direct tradeoff between increasing irrigation upstream and impacts on the wetlands downstream. Scenario 3, which yields the lowest upstream irrigation gains, also has the least impact in terms of floodplain losses, whereas Scenario 5 has both the highest irrigation gains and floodplain losses.

The results confirm that in all the scenarios simulated the additional value of production from large-scale irrigation schemes does not replace the lost production attributed to the wetlands downstream. Gains in irrigation values account for at most around 17% of the losses in floodplain benefits.

This combined hydrological-economic analysis would suggest that no new upstream developments should take place in addition to Tiga Dam. Moreover, a comparison of Scenario 3 to Scenario 2 in the analysis shows that it is economically worthwhile to reduce floodplain losses through releasing a substantial volume of water during the wet season, even though this would not allow Tiga Dam to supply the originally planned 27 000 ha on KRIP-I.

Although Scenario 3 is the preferred scenario, it is clearly unrealistic. As indicated above, Challawa Gorge was completed in 1992, and in recent years several small dams have been built on the Hadejia's tributaries while others are planned. Thus Scenario 4 most closely represents the current situation, and Scenario 5 is on the way to being implemented - although when the construction of Kafin Zaki Dam might occur is presently uncertain. As indicated in Table 3.b., full implementation of all the upstream dams and large-scale irrigation schemes would produce the greatest overall net losses, around US$20.2-20.9 million.

These results suggest that the expansion of the existing irrigation schemes within the river basin is effectively 'uneconomic'. The construction of Kafin Zaki Dam and extensive large-scale formal irrigation schemes within the Jama'are Valley do not represent the most appropriate developments for this part of the basin. If Kafin Zaki Dam were to be constructed and formal irrigation within the basin limited to its current extent, the introduction of a regulated flooding regime (Scenario

6) would reduce the scale of this negative balance substantially, to around US$15.4-16.5 million. The overall combined value of production from irrigation and the floodplain would however still fall well below the levels experienced if the additional upstream schemes were not constructed.[9]

Such a regulated flooding regime could also produce additional economic benefits that are not captured in our analysis. Greater certainty over the timing and magnitude of the floods may enable farmers to adjust to the resulting reduction in the risks normally associated with floodplain farming. Enhanced dry season flows provided by the releases from Challawa Gorge and Kafin Zaki dams in Scenario 6 would also benefit farmers along the Hadejia and Jama'are Rivers while the floodplain's fisheries may also experience beneficial impacts from the greater extent of inundation remaining throughout the dry season. The introduction of a regulated flooding regime for the existing schemes within the basin may be the only realistic hope of minimizing floodplain losses. Proposed large-scale schemes, such as Kafin Zaki, should ideally be avoided if further floodplain losses are to be prevented. If this is not possible the designs for water resource schemes should enable the release of regulated floods in order to, at least partly, mitigate the loss of floodplain benefits which would inevitably result.

Currently, as a result of such economic and hydrological analyses of the downstream impacts of upstream water developments in the Hadejia-Jama'are floodplain both the States in Northern Nigeria and the Federal Government have become interested in developing regulated flooding regimes for the upstream dams, and have been reconsidering the construction of Kafin Zaki Dam.

Dynamic Models of Ecological Services Supporting Economic Activity

The production function approach can also be incorporated into intertemporal models of renewable resource harvesting in cases where the ecological function affects the growth rate of a stock over time. In such cases, the production function link is a dynamic one, as the ecological function affects the rate at which a renewable resource increases over time, which in turn affects the amount of offtake, or harvest, of the resource. The basic approach to valuation of an environmental input to renewable resource production in a dynamic context is outlined by Barbier and Strand (1998), Ellis and Fisher (1987), Freeman (1993), Hammack and Brown (1974), Kahn and Kemp (1987), McConnell and Strand (1989), and Swallow (1990) and (1994).

As shown by Barbier and Strand (1998), adapting bioeconomic fishery models to account for the role of a mangrove system in terms of supporting the fishery as a breeding ground and nursery habitat is fairly straightforward, if it is assumed in the fishery model that the effect of changes in mangrove area is on the carrying capacity of the stock and thus indirectly on production.[10] Defining X_t as the stock of fish measured in biomass units, any net change in growth of this stock over time can be represented as

$$X_{t+1} - X_t = F(X_t, M_t) - h(X_t, E_t), \quad F_X > 0,\ F_M > 0. \qquad \textbf{(9)}$$

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

Equation (9) can now be employed in a standard intertemporal harvesting model of the fishery, where depending on the management regime, harvesting over time can either be depicted to occur under open access conditions (i.e. effort in the fishery adjusts over time to the availability of profits) or under optimal management conditions (the discounted net returns from harvesting the fishery are maximized over time). The effect of a change in mangrove area can therefore be valued in terms of changes in the optimal path of harvesting over the period of analysis and in terms of the changes in the long run equilibrium of the fishery.

Figure 2 shows the fairly straightforward case analyzed by Barbier and Strand, where the effects of a change in mangrove area is depicted in terms of influencing the long run equilibrium of an open access fishery. In the figure, the long run equilibrium of the fishery is depicted in terms of steady values for effort, E, and fish stocks, X. As discussed above, the carrying capacity of the fishery is assumed to be an increasing function of mangrove area, i.e. $K = K(M)$, $K_M > 0$. Trajectory one shows an optimal path to a stable long-run equilibrium for the fishery. In this case, a decrease in mangrove area causes the long-run level of fishing effort to fall. As harvesting levels are generally positively related to effort levels, the consequence of mangrove deforestation is also a decrease in equilibrium fish harvest.

Case study: mangrove-fishery linkages, Campeche, Mexico[11]

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

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

Barbier and Strand model the effects of mangrove deforestation in Laguna de Terminos by use of comparative static analysis of the long run equilibrium, as depicted in Figure 2. In their model of the Campeche shrimp fishery, they assume that the basic growth function of the fishery is logistic and that shrimp harvesting follows the conventional Schaefer production process, $h_t = qE_tX_t$. Thus (9) becomes

$$X_{t+1} - X_t = [r(K(M_t) - X_t) - qE_t]X_t, \qquad \textbf{(10)}$$

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

To estimate the comparative static effects of a chance in mangrove area on long run shrimp harvesting, Barbier and Strand assume a proportional relationship between mangrove area and carrying capacity, i.e. $K(M) = \alpha M,\ \ \alpha > 0$. As the shrimp stock is constant in the long run equilibrium, $X_t = X_{t+1} = X$, then using this condition in (10) and the Schaefer production function to substitute for X, the following relationship between shrimp production, mangrove area and effort is derived

$$h = qEK(M) - \frac{q^2}{r}E^2 = q\alpha EM - \frac{q^2}{r}E^2. \qquad \textbf{(11)}$$

The authors estimate equation (11) by employing 1980-90 time series data on shrimp harvests, effort and mangrove area for Campeche, Mexico to derive the parameters $b_1 = \alpha q$ and $b_2 = - q^2/r$.

A second condition of the long run equilibrium of an open access fishery is that profits will be zero, i.e. $ph = cE$, where p is the price of shrimp catch and c is the cost of fishing effort. In order to simulate the comparative static effects of a change in mangrove area, Barbier and Strand assume that this 'zero profit' condition holds for the Campeche shrimp fishery. Using actual price data on shrimp catches over this period, they calculate the costs of effort, c^A, necessary for the zero profit condition to hold for the Campeche fishery over 1980-90. Using the estimated parameters of equation (11) with the price and cost data, the authors simulate the effects of a change of mangrove area on equilibrium harvesting and gross revenues in the Campeche shrimp fishery over 1980-90.

The results are depicted in Table 4. On average over the 1980-90 period, a marginal (in km^2) decline in mangrove area produces a loss of about 14.4 metric tons of shrimp harvest and nearly US$140,000 in revenues from the Campeche fishery each year. However, given the relatively small rate of annual mangrove deforestation in the region over the 1980-90 period - around 2 km^2 per year - the resulting loss in shrimp harvest and revenues does not appear to be substantial - only around 0.4% per year.

The simulation in Table 4 also demonstrates how the economic losses associated with mangrove deforestation are affected by long-run management of the open access fishery. As noted above, the early years of the period of analysis (e.g. 1980-81) were characterized by much lower levels of fishing effort and higher harvests (e.g. on average around 4,800 combined vessels extracting about 8.5 KMT annually). Table 4 shows that, if this earlier period represented the open access equilibrium of the fishery, the economic impacts of a marginal (km^2) decline in mangrove area would be a reduction in annual shrimp harvests of around 18.6 tons, or a loss of about US$153,300 per year. In contrast, the last two years of the analysis (e.g. 1989-90) saw much higher levels of effort and lower harvests in the fishery (e.g. around 6,700 combined vessels extracting 5.3 KMT annually). As a consequence, if this latter period represents the open access equilibrium, then a marginal decline in mangrove area would result in annual losses in shrimp harvests of 8.4 tons, or US$86,345 each year.

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

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

Conclusion

Incorporating environmental values in investment decisions in developing countries will continue to be an important issue. As many environmental resources and services in these countries either directly or indirectly contribute to economic activity, the production function approach to valuing this contribution will continue to be an essential methodology.

In recent years there have several developments in the use of this approach. This paper has reviewed three key applications to resource management problems in developing countries. The case study of gum arabic in Sudan shows that it is important to consider both the wider environmental benefits of a production system rather than just its marketed output. The case study of water diversion in Northern Nigeria illustrates the need to model and analyse the downstream hydrological impacts of water diverted upstream. Finally, the case study of mangrove-fishery linkages in Campeche, Mexico indicates how the production function approach can be extended to a dynamic analysis of the impacts of mangrove deforestation on an offshore shrimp fishery.

There are of course many more applications of the production function approach to environmental valuation in developing countries. As this approach is developed and applied more frequently to a wide range of problems, its usefulness as a project appraisal and policy analysis tool should hopefully increase.

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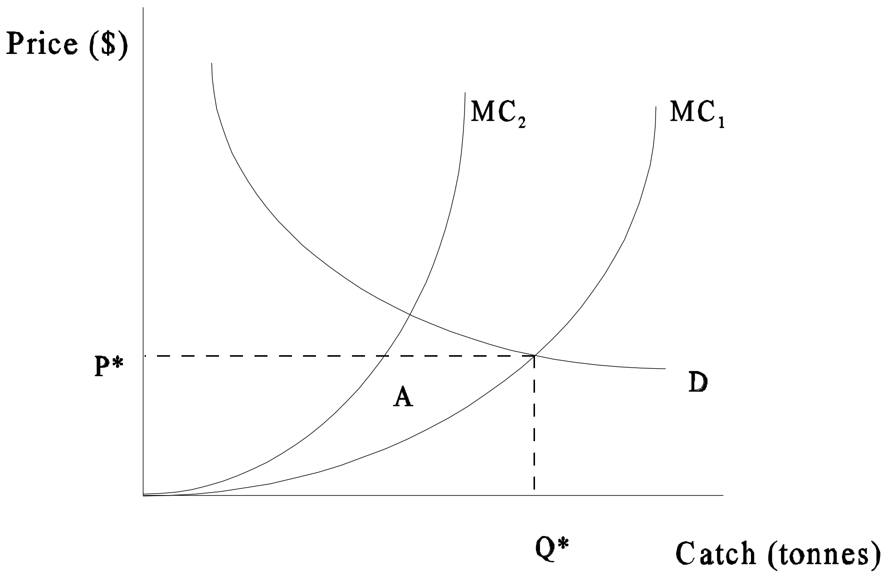
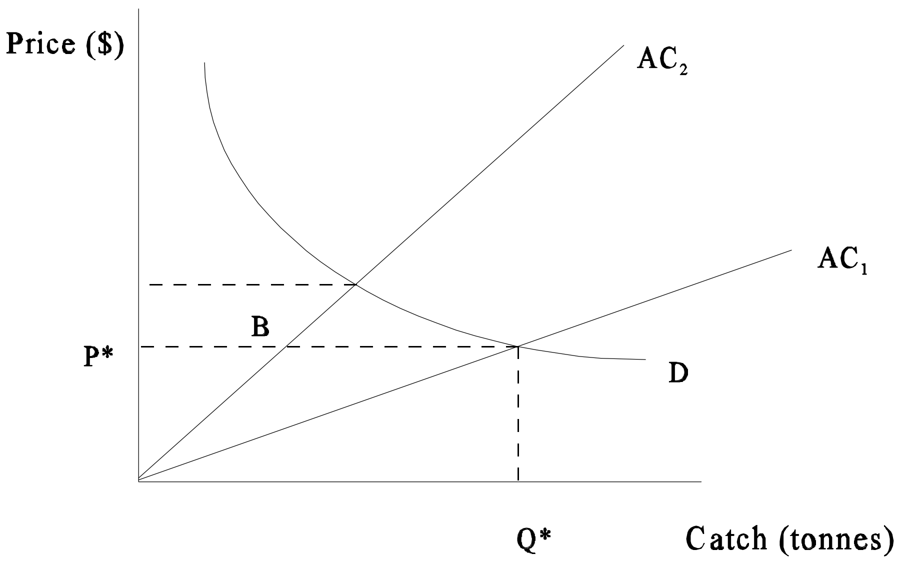
Figura 1. Welfare Measures in Optimally Managed and Open Acces Fisheries



The welfare impact of a change in wetland area on an optimally managed fishery is the change in consumer and producer surplus (area A).



The welfare impact of a change in wetland area on an open access fishery is the change in consumer surplus (area B).

Table 1. **Economic Analysis of Six Cropping Systems, Sudan**

(1989/90 Sudanese Pounds per Feddan, 10% Discount Rate)

	Acacia Senegal 1/	Sorghum	Millet	Groundnuts	Sesame	**ALL CROPS**
BN	2989.57	4606.83	2931.58	--	--	**10527.98**
WN	3923.82	- 432.99	486.30	5330.92	8605.12	**17913.17**
NK	1471.36	--	2091.77	--	6020.94	**9584.07**
SK	882.99	1644.44	--	13109.44	13776.95	**29413.82**
ND	1240.87	--	2363.89	9687.69	--	**13292.45**
SD	1884.02	3830.80	3775.60	9715.61	--	**19206.03**
ALL	**2065.44**	**2412.27**	**2392.83**	**9460.92**	**9467.67**	**16656.25**

Notes: BN = Blue Nile Province, clay soils, largeholder
WN = White Nile Province, sandy soils, smallholder
NK = North Kordofan Province, sandy soils, smallholder
SK = South Kordofan Province, clay soils, largeholder
ND = North Darfur Province, sandy soils, smallholder
SD = South Darfur Province, sandy soils, smallholder
ALL = Average of all six systems

1/ Total NPV from gum, fuelwood and fodder, except for Blue Nile and South Kordofan (gum only).

Source: Barbier (1992)

Table 2. Scenario for Upstream Projects in the Hadejia-Jama are River Basin, Northern Nigeria

Scenario (Time Period)	Dams	Regulated Releases ($10^6 m^3$)	Irrigation Schemes
1 (1974-1985)	Tiga	Naturalised Wudil flow (1974-1985)	No KRIP-I
1a (1974-1990)	Tiga	Naturalised Wudil flow (1974-1990)	No KRIP-I
2 (1964-1985)	Tiga	None	KRIP-I at 27 000 ha
3 (1964-1985)	Tiga	400 in August for sustaining floodplain	KRIP-I at 14 000 ha
4 (1964-1985)	Tiga	None	KRIP-I at 27 000 ha
	Challawa Gorge	348 yr^{-1} for HVP	
	Small dams on Hadejia tributaries		
5 (1964-1985)	Tiga	None	KRIP-I at 27 000 ha
	Challawa Gorge	348 yr^{-1} for HVP	
	Small dams on Hadejia tributaries		
	Kafin Zaki		
	HVP	None	84 000 ha
		None	12 500 ha
6 (1964-1985)	Tiga	350 in August	KRIP-I at 14 000 ha
	Challawa Gorge	248 yr^{-1} and 100 in July	
	Small dams on Hadejia tributaries		
	Kafin Zaki		
	HVP	100 per month: Oct-Mar and 550 in August	None
		Barrage open in August	8 000 ha

Notes: KRIP-I = Kano River Irrigation Project Phase I
HVP = Hadejia Valley Project

Source: Barbier and Thompson (1998).

Table 3a. Impact of Scenarios on Mean Peak Flood Extent and Gains in Total Irrigated Area

	Scenario 1	Scenario 1a	Irrigated Area (km^2)
Scenario 2	-150.62	-211.20	270
Scenario 3	-95.25	-55.83	140
Scenario 4	-265.02	-325.60	270
Scenario 5	-870.49	-931.07	1 235
Scenario 6	-574.67	-635.25	220

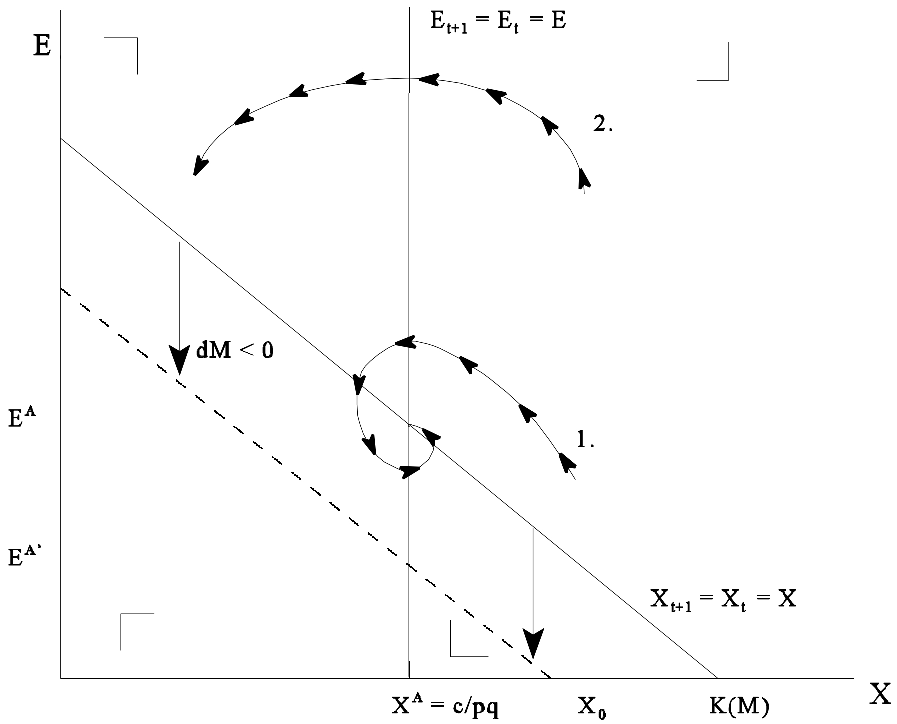
Table 3b. Impact of Scenarios in Terms of Losses in Floodplain Benefits versus Gains in Irrigated Production, Net Present Value (US$ 1989/90 Prices)

		Scenario 1			Scenario 1a		
	Irrigation Value [1] a/	Floodplain Loss [2] b/	Net Loss [2] - [1]	[1] as % of [2]	Floodplain Loss [3] b/	Net Loss [3] - [1]	[1] as % of [3]
Scenario 2	682 983	-4 045 024	3 362 041	16.88	-5 671 973	-4 988 990	12.04
Scenario 3	354 139	-2 558 051	-2 203 912	13.84	-4 184 999	-3 830 860	8.46
Scenario 4	682 963	-7 117 291	-6 434 328	9.60	-8 744 240	-8 061 277	7.81
Scenario 5	3 124 015	-23 377 302	-20 253 287	13.36	-24 004 251	-20 880 236	13.01
Scenario 6	556,505	-15 432 952	-14 876 447	3.61	-17 059 901	-16 503 396	3.26

Notes: a/ Based on the mean of the net present values of per hectare production benefits for the Kano River Irrigation Project Phase I (see Barbier and Thompson 1998), and applied to the gains in total irrigation area shown in Table 3a.

b/ Based on the mean of the net present values of total benefits for the Hadejia-Jama'are floodplain (see Barbier and Thompson 1998), averaged over the actual peak flood extent for the wetlands of 112 817 ha in 1989/90 and applied to the differences in mean peak flood extent shown in Table 3a.

Figure 3. Mangrove Loss and the Long Run Equilibrium of an Open Access Fishery



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

Source: Barbier and Strand (1998).

Table 4. **Simulation Results for the Effects of Mangrove Loss on the Open Access Equilibrium of the Campeche Shrimp Fishery, 1980-90**

Parameter Estimates:

$b_1 = 4.4491$
$b_2 = -0.4297$

Simulation Estimates of a Marginal Change in Mangrove Area (d*M*)

Year	Price (p) US$/kg a/	Cost (c^A) US$/vessel b/	Change in Equilibrium Harvest (dh^A) metric tons	Change in Equilibrium Revenues (pdh^A) US$	Change %
1980	7.10	13,984	20.40	144,808	0.23
1981	9.68	15,628	16.72	161,826	0.20
1982	10.57	13,816	13.53	143,060	0.18
1983	9.80	13,636	14.41	141,197	0.18
1984	9.83	14,096	14.85	145,963	0.19
1985	9.80	16,687	17.63	172,798	0.20
1986	10.00	15,013	15.55	155,460	0.19
1987	10.22	14,363	14.55	148,731	0.20
1988	10.56	14,132	13.86	146,334	0.20
1989	10.21	10,000	10.14	103,547	0.17
1990	10.40	6,677	6.65	69,143	0.14
Mean	**9.83**	**13,457**	**14.39**	**139,352**	**0.19**

Notes: a/ US$/kg, in real (1982) prices.

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

Source: Barbier and Strand (1998).

Notes

1. It is assumed that all costs and benefits are discounted at some positive rate into present value terms.

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

3. The following case study is based on Barbier (1992).

4. The assumption that there is no cost of using the supply of water available to the agricultural system is certainly realistic for a floodplain system dependent on the natural recession of flood water as a source of irrigation. In the case of irrigation provided by a human-made reservoir and channel network, the assumption of a freely available supply suggests that the fixed costs of the water reservoir and network were absorbed by an external agency (e.g. central or regional government), and there is no recurrent charge to the irrigated agricultural system for using water as an input. Obviously, this raises issues over the efficiency of water input use, and the possibility over time of excessive use relative to the supply of available water. Although an extremely important issue, particularly with regard to the supply of irrigated water (see Repetto 1986 for a discussion and examples),this problem is not an explicit focus of this paper. The problem could easily be incorporated into the model by examining how the amount of water abstracted for irrigation in each time period, $k(t)$, influences the rate of depletion of the total downstream water stock available to irrigation, $dW/dt = - k(t)W(t)$, and assuming a cost of abstraction $c_w \geq 0$.

5. In what follows, for simplicity, the fixed costs of establishing the upstream water project and irrigation will also be ignored. See also the previous note on how water abstraction costs could be included.

6. Although it is possible that the optimal rate of diversion, d^*,may lead to complete depletion of the stock of water available downstream, W, over the planning horizon $(0,T)$. It is also possible that d^* may also be zero as $t \rightarrow T$. The inclusion of cumulative diversion into the upstream irrigation network, $\int d\ dt = W(0) - W(t)$, as an argument in h_2 suggests that output from the upstream agricultural project does not necessarily have to fall to zero if $d^* = 0$. It is fairly easy to work out the conditions determining the optimal path of $d(t)$ as well as the rate of change, dd^*/dt, from the first order conditions. See Barbier (1994) and Dasgupta and Heal (1979) for further discussion.

7.The references for this case study are Barbier and Thompson (1998). See also Barbier, Adams and Kimmage (1993).

8. Note that one reason for these high losses in floodplain benefits is that the total production area dependent on the wetlands is around 6.5 times greater than the actual area flooded. This critical feature of a semi-arid floodplain, its ability to 'sustain' a production area much greater than the area flooded, is often underestimated and ignored. This in turn means that changes in flood extent have a greater multiplier impact in terms of losses in economic benefits in production areas within and adjacent to the floodplain, because of the high dependence of these areas on regular annual flooding. See Barbier and Thompson (1998) for more details.

9. Some of the upstream water developments are being used or have the potential to supply water to Kano City. Although these releases are included in the hydrological simulations, the economic analysis was unable to calculate the benefits to Kano City of these water supplies. However, the hydrological analysis shows that the proposed regulated water release from Tiga Dam to reduce downstream floodplain losses would not affect the ability of Tiga Dam to supply water to Kano. Although the potential exists for Challawa Gorge to supply additional water to Kano, it is unclear how much water could be used for this purposed. The resulting economic benefits are unlikely to be large enough to compensate for the substantial floodplain losses incurred by the Gorge and the additional upstream developments in the Hadejia Valley. Currently, there are no plans for Kafin Zaki Dam to be used to supply water to Kano. In addition, the economic analysis was unable to calculate other important floodplain benefits, such as the role of the wetlands in supporting pastoral grazing and in recharging groundwater both within the floodplain and in surrounding areas. Groundwater recharge by the floodplain may provide potable water supplies to populations within the middle and lower

parts of the river basin, and supply tubewell irrigation for dry season farming downstream (Barbier *et al.*1993).

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

11. This case study draws on Barbier and Strand (1998).