HABITAT–FISHERY LINKAGES AND MANGROVE LOSS IN THAILAND

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The article develops a dynamic model of habitat–fishery linkage in which the habitat is being converted. The basic model is applied to a case study of the impacts of mangrove deforestation on the artisanal marine demersal and shellfish fisheries in Thailand. The comparative static effects of a change in mangrove area on the long-run equilibrium level of effort and fish stocks, as well as on the resulting market harvesting supply of the fishery, are determined. By estimating parameters through pooled time-series and cross-sectional data over the 1983–96 period for the five coastal zones along the Gulf of Thailand and Andaman Sea, the welfare impacts of mangrove deforestation are estimated. Mangrove conversion is expected to be a function of the return to shrimp farming and the input costs to farming shrimp, plus exogenous economic factors. The resulting aggregate reduced-form level of mangrove clearing by all farmers in coastal areas is empirically estimated across the five coastal zones in Thailand over 1983–96. The policy implications of the findings are discussed with respect to Thailand and the modeling of habitat–fishery linkages. (*JEL* O13, Q22, Q23, C61, C23)

I. INTRODUCTION

An extensive literature in ecology has emphasized the role of coastal wetland habitats in supporting neighboring marine fisheries (Mitsch and Gosselink, 1993; Mooney et al., 1995; World Conservation Monitoring Centre [WCMC], 1992; World Resources Institute [WRI], 1996; Yañez-Arancibia and Day, 1988). The ecological function of particular interest is the role of mangrove or coastal estuarine wetland systems in serving as a breeding ground or nursery for offshore fisheries. The main concern expressed by ecologists is that as coastal wetlands around the world become increasingly degraded, converted, or overexploited, the resulting loss in fish nurseries and breeding grounds will have a significant impact on marine fisheries (WCMC, 1992). Although there are many reasons for the destruction of mangrove forests—including increasing population pressure, coastal development, mining, conversion to salt ponds and agriculture, and overharvesting of the forests—the largest factor in recent years has been the widespread expansion of aquaculture ponds

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ABBREVIATIONS

GDP: Gross Domestic Product
GLS: Generalized Least Squares
KMT: Thousand Metric Tons
LM: Lagrange Multiplier
LR: Likelihood Ratio
OLS: Ordinary Least Squares
WCMC: World Conservation Monitoring Centre
WRI: World Resources Institute

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into mangrove forests globally (Aksornkoae et al., 1986; Spalding et al., 1997; WRI, 1996). The conversion of mangroves to shrimp farming has been particularly evident in Thailand over the past 25 years.

Because of the potential welfare implications arising from the decline in coastal breeding habitats and nurseries, such as mangrove systems, economists have also become interested in valuing the production externality arising from this ecological support function. Recent attempts to value mangrove habitat–fishery linkages in Mexico, Indonesia, and Thailand indicate that mangrove deforestation is contributing to fisheries decline and significant welfare losses (Barbier and Strand, 1998; Ruitenbeek, 1994; Sathirathai and Barbier, 2001). In addition, related studies have attempted to develop a methodology for assessing habitat–fishery problems more generally, such as analyzing the competition between mangroves and shrimp aquaculture in Ecuador (Parks and Bonifaz, 1994); determining the value of a multiple-use mangrove system under different management options in Bintuni Bay, Irian Jaya, Indonesia (Ruitenbeek, 1994); and examining general coastal system trade-offs, such as the effects of development and/or pollution on habitat–fishery linkages (Kahn and Kemp, 1985; McConnell and Strand, 1989; Strand and Bockstael, 1990; Swallow, 1990, 1994).

Many previous studies of the impacts of coastal developments on habitat–fishery linkages have generally involved a static rather than a dynamic approach (Barbier, 2000). In static approaches, no attempt is made to model either the dynamic influences of a change in a coastal habitat on the growth in the biological populations of the fishery or to include the effects of habitat loss through coastal development over time (Ellis and Fisher, 1987; Freeman, 1991; Sathirathai and Barbier, 2001). Instead, a single-period model of fishery production is constructed, whereby the coastal habitat, such as mangrove area, is assumed to be one of the inputs along with fishing effort. The welfare contribution of changes in this habitat input is then determined through producer and consumer surplus measures of any corresponding changes in the market equilibrium for harvested fish. For example, in Thailand, using such a static analysis, the welfare losses associated with the impacts of mangrove deforestation on coastal fisheries in Surat Thani province were recently estimated to be around US $21 to $52 per ha (Sathirathai and Barbier, 2001).

The article hopes to make several important contributions to this growing literature on habitat–fishery linkages as well as analyzing the problem of mangrove deforestation in Thailand. The key feature of the approach is to analyze simultaneously the two aspects of habitat–fishery linkage in coastal tropical areas: (1) the impacts of shrimp farming on mangrove systems and (2) the corresponding effects of mangrove loss on habitat–fishery linkages. The article approaches this analysis in several stages.

First, the author develops a dynamic model of habitat–fishery linkage in which the habitat is being converted. The basic model is applied to a case study of the impacts of mangrove deforestation on the artisanal marine demersal and shellfish fisheries in Thailand. The comparative static effects of a change in mangrove area on the long-run equilibrium level of effort and fish stocks, as well as on the resulting market harvesting supply of the fishery, are determined. By estimating parameters through pooled time-series and cross-sectional data over the 1983–96 period for the five coastal zones along the Gulf of Thailand and Andaman Sea, it is possible to determine explicitly the welfare impacts on Thailand's coastal fisheries of mangrove deforestation.

Second, the article also develops a model of the mangrove clearing decisions of shrimp farmers under open-access conditions. Mangrove conversion is expected to be a function of the returns and input costs to shrimp farming plus exogenous economic factors. The resulting aggregate reduced-form level of mangrove clearing by all farmers in coastal areas is empirically estimated across the five coastal zones in Thailand over 1983–96.

Finally, both the habitat–fishery model and the mangrove deforestation model are linked to estimate the welfare effects in Thailand's coastal fisheries resulting from changes in mangrove areas. These welfare losses are then compared to the annual economic value of shrimp production in Thailand. A simulation is also conducted to determine the impacts of the recent economic crisis in Thailand—exchange rate devaluation and falling gross domestic product (GDP)—on the welfare losses imposed by mangrove deforestation on coastal fisheries.

The structure of the article is as follows. The next section provides a brief background to the problem of mangrove deforestation and shrimp aquaculture expansion in Thailand's coastal provinces. Section III develops the dynamic model of habitat–fishery linkage and estimates its key parameters through a pooled time-series and cross-sectional analysis for Thailand's five coastal zones. Section IV develops the model of mangrove clearing decisions by shrimp farmers under open-access conditions and presents the estimation of the resulting reduced-form relation for mangrove conversion across Thailand's five coastal zones. Section V links the two models to examine the welfare impacts of mangrove deforestation and simulations of the effects of the economic crisis in Thailand. Section VI discusses the policy implications of the analysis for Thailand and its general relevance to the modeling of habitat–fishery linkages.

II. BACKGROUND

In Thailand, the conversion of coastal land for commercial shrimp farming is a highly debated and controversial topic. In the late 1990s, the total value of export earnings for shrimp was around \$1–\$2 billion annually, and the government has been keen to expand these exports (Jitsanguan et al., 1999; Tokrisna, 1998). Since 1979, Thailand has been the world's major shrimp producer, and one-third of all shrimp marketed internationally is from Thailand. Although shrimp are also caught in coastal fisheries, the vast majority of Thailand's shrimp production now comes from aquaculture. As a consequence, expansion of shrimp production and exports has caused a major transformation in coastal areas.

Thailand's coastline is vast, stretching for 2,815 km, of which 1,878 km is on the Gulf of Thailand and 937 km on the Andaman Sea (Indian Ocean) (Kaosa-ard and Pednekar, 1998). In recent years, the expansion of intensive shrimp farming in the coastal areas of southern Thailand has led to rapid conversion of mangroves (Coastal Research Institute, 1995). Over 1975–93 the area of mangroves virtually halved, from 312,700 to 168,683 ha. Although the rate of mangrove deforestation has slowed more recently, throughout the 1980s and 1990s the annual loss has averaged around 5,000 ha/year.

Mangrove conversion for shrimp aquaculture began in Thailand as early as 1974. However, this process was accelerated in 1985, when the increasing demand for shrimp in Japan pushed up the border-equivalent price to \$100 per kg (Bantoon, 1994). For example, over the 1981–85 period in Thailand, annual shrimp production through aquaculture was around 15 thousand metric tons (KMT), but by 1991 it had risen to over 162 KMT and by 1994 to over 264 KMT (Kaosa-ard and Pednekar, 1998). Although it has declined slightly in recent years, production still remains over 210 KMT.

Shrimp farm area has expanded from 31,906 ha to 66,027 ha between 1983 and 1996. A more startling figure is the increase in the number of farms during that period: from 3,779 to 21,917. In general, this reflects a rapid shift from more extensive to more small-scale, intensive and highly productive aquaculture systems of on average two or three ponds with each pond comprising up to 1 ha in size (Kongkeo, 1997; Tokrisna, 1998). However, much of the semi-intensive and intensive shrimp farming in Thailand is short-term and unsustainable, that is, water quality and disease problems mean that yields decline rapidly and farms are routinely abandoned after five to six years of production (Dierberg and Kiattisimkul, 1996; Flaherty and Karnjanakesorn, 1995; Thongrak et al., 1997; Tokrisna, 1998). Shrimp farm expansion has slowed in recent years, but unsustainable production methods and lack of knowhow have meant that more expansion takes place every year simply to replace unproductive and abandoned farms. Estimates of the amount of mangrove conversion due to shrimp farming vary, but recent studies suggest that up to 50%–65% of Thailand's mangroves have been lost to shrimp farm conversion since 1975 (Dierberg and Kiattisimkul, 1996; Tokrisna, 1998). In provinces close to Bangkok, such as Chanthaburi, mangrove areas have been devastated by shrimp farm developments (Raine, 1994). More recently, Thailand's shrimp output has been maintained by the expansion of shrimp farming activities to the far southern and eastern parts of the Gulf of Thailand and across to the Andaman Sea coast (Flaherty and Karnjanakesorn, 1995; Sathirathai, 1998).

It is possible to design shrimp aquaculture systems in coastal areas that do not involve removal of vegetation and areas naturally fed by tidal conditions. In addition, the water pollution generated by shrimp farming can be minimized, provided that wastewater from the farm has been treated before being released, However, the establishment of these "sustainable" farm systems is too expensive for the type of small-scale pond operations found in much of Thailand, which are dependent on highly intensive and untreated systems through rapid conversion of mangrove and coastal resources (Thongrak et al., 1997; Tokrisna, 1998). Much of the financial investment in coastal shrimp farms is from wealthy individual investors and business enterprises from outside of the local community (Flaherty and Karnjanakesorn, 1995). Although some hiring of local labor occurs, in the past shrimp farm owners have tended to hire Burmese workers because their wage rates are much lower.

Ill-defined property rights have also contributed to the rapid conversion of mangroves to shrimp farms. Although the state through the Royal Forestry Department ostensibly owns and controls mangrove areas, in practice they are de facto open-access areas onto which anyone can encroach. Mangrove deforestation in southern Thailand has also been affected by the increase in tourist infrastructure, agricultural expansion, and urban developments in coastal areas (Dierberg and Kiattisikul, 1996; Tokrisna, 1998). Until recently, government policy considered mangroves as areas of wasteland that could be freely reclaimed for development and highly profitable economic activities (Sathirathai, 1998). However, there is now increasing concern that the loss of mangroves may have devastating welfare effects for local coastal communities, particularly those dependent on products collected from mangrove forests and on coastal artisanal fisheries (Sathirathai and Barbier, 2001).

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 makes up a separate fifth zone.[1] The mangroves along these coastal zones of southern Thailand 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.[2] The artisanal marine fisheries of the five major coastal zones of Thailand are in turn largely dominated by shellfish and also depend on demersal fish. For example, in 1994 shrimp, crab, squid, and cuttlefish alone accounted for 67% of all catches 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 2,500 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. Though 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. In recent years, a growing number of traditional coastal fishing households have left the industry and been replaced by new entrants, such as recent migrants from the poorer north and northeast regions of Thailand, Myanmar, Cambodia, and Laos. Although a license 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

1. 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 Andaman Sea consists of the following coastal provinces: Ranong, Phangnga, Phuket, Krabi, Trang, and Satun (zone 5).

2. 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.

for supporting community-based fishery management (Kaosa-ard and Pednekar, 1998).

Although the coastal artisanal fisheries account for 78% of total marine establishments in Thailand, they contribute only 6% of the total marine production by volume and 19% by value. However, these figures under-represent the actual value of the coastal zone fisheries to Thailand (Kaosa-ard and Pednekar, 1998). First, it is estimated that about one-third of the marine landings attributed to Thailand's formal fishing sector are caught illegally by commercial trawlers outside of Thai territorial waters. Second, an unknown amount of fish is also caught illegally by commercial operators within the 3 km coastal zone that is supposed to be reserved exclusively for small-scale fishers. Finally, due to the lack of regulation of the coastal artisanal fisheries, many small-scale fishing boats are unregistered, and the total catch taken in coastal waters is thought to be much greater than indicated by the official figures attributed to the small-scale fishing sector.

III. A BIOECONOMIC MODEL OF MANGROVE–FISHERY LINKAGES

This section introduces a dynamic model depicting the role of mangroves in supporting the artisanal shellfish and demersal fisheries of Thailand's 3 km coastal zone waters. That is, an intertemporal model of an open-access fishery is adopted to allow for the possible effect of changes in mangrove area on the carrying capacity of the stock and thus indirectly on production. However, given that any change in the large volume of harvests produced in the Thai fisheries will invariably affect the market price of fish, the model allows for such price effects in the analysis of mangrove–fishery linkages by assuming that these open access fisheries face a finite elasticity of demand.

The standard approach for incorporating habitat–fishery linkages in bioeconomic fishery models is to assume that the role of a mangrove system in terms of supporting the fishery as a breeding ground and nursery habitat affects the carrying capacity and thus biological growth of the stock.[3] 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

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

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, $h(X_t, E_t)$, which is a function of the stock as well as fishing effort, E_t. The influence of mangrove area, M_t, as a breeding ground and nursery habitat on growth of the fish stock is assumed to be positive, that is, $\partial F/\partial M_t = F_m > 0$, as an increase in mangrove area will mean more carrying capacity for the fishery and thus greater biological growth. The standard assumption that the harvesting function is concave with respect to its two arguments also applies.

As discussed in the previous section, the artisanal fisheries of Thailand are essentially open access. Following standard analysis, this suggests that fishing effort next period will adjust in response to the real profits made in the current period (Clark, 1976). Letting $p(h)$ represent landed fish price per unit harvested, c the real unit cost of effort, and $\phi > 0$ the adjustment coefficient, then the fishing effort adjustment equation is

$$(2) \quad E_{t+1} - E_t = \phi[p(h)h(X_t, E_t) - cE_t], \quad p_h < 0.$$

Any increase in harvest is assumed to lead to a fall in price, p.

The two-equation system constitutes the basic dynamic model for analyzing fishery–mangrove linkages in Thailand. The analysis of these linkages is conducted by examining the effects of a change in mangrove area on the long-run open-access equilibrium of the fishery. In this steady state, both the fish stock and the level of fishing effort are assumed to be constant over time: $X_{t+1} = X_t = X$ and $E_{t+1} = E_t = E$. It is also assumed initially that mangrove area is in equilibrium, $M_t = M_{t+1} = M$. This suggests that the open-access equilibrium can be represented by the following equations, and solved in terms of the steady-state levels of stock, X, and effort, E

3. See Barbier (2000) for a discussion and review of the various approaches employing this assumption.

(and a constant M):

$$(3) \qquad F(X,M) - h(X,E) = 0, \quad \text{for} \quad X_{t+1} = X_t = X, \quad M_{t+1} = M_t = M$$

$$(4) \qquad P(h)h(X,E) - cE = 0, \quad \text{for} \quad E_{t+1} = E_t = E.$$

Equation (3) indicates the combinations of fishing effort and stock size (and thus also mangrove area) that will lead to a constant level of stock in the long run. Equation (4) is the standard open-access condition that any profits in the fishery will be competed away in the long run.

Equilibrium conditions (3) and (4) can be solved by assuming a conventional bioeconomic fishery model with biological growth characterized by a logistic function, $F(X,M) = rX(K(M) - X)$, and harvesting by a Schaefer production process, $h = qXE$, where q is a catchability coefficient, r is the intrinsic growth rate, and $K(M)$, $K_M > 0$, is the impact of mangrove area on carrying capacity, K. Substituting these expressions into (3) and (4) yield

$$(5) \quad E = [r(K(M) - X)]/q, \quad \text{for } X_{t+1} = X_t = X, \quad M_{t+1} = M_t = M;$$

$$(6) \quad X = c/[p(h)q], \quad \text{for } E_{t+1} = E_t = E.$$

This two-equation equilibrium system can be depicted graphically in $X - E$ space. Totally differentiating both equations, and using the elasticity of demand $\varepsilon = p(h)/p_h h(X,E)$ as well as (13), yields

$$(7) \quad -rdX - qdE + rK_M dM = 0, \quad \text{and } dE/dX = -r/q < 0;$$

$$(8) \quad [(1+\varepsilon)qX - (c\varepsilon)/[p(h)]]\,dE + (1+\varepsilon)qE\,dX = 0 \quad \text{and}$$
$$dE/dX = -[(1+\varepsilon)qE]/[c/p(h)] \gtreqless 0 \text{ if } |\varepsilon| \lesseqgtr 1.$$

Figure 1 illustrates the long-run equilibrium (X^A, E^A) for the open-access fishery corresponding to the elastic demand case.

Note that the determinant of the matrix of coefficients, D, of the system represented by (7) and (8) has the following properties

$$(9) \quad D = (1+\varepsilon)qE - (cr)/[p(h)] < 0 \quad \text{if } |\varepsilon| > 1 \text{ or } |\varepsilon| < 1 \text{ and } r/q > [(1+\varepsilon)qE]/[cp(h)].$$

$D < 0$ is a necessary condition for a saddle-point, which is assumed to be the case for the equilibrium depicted in Figure 1.

Using Cramer's rule, the comparative static effects on equilibrium stock and effort levels (X^A, E^A) of a change in mangrove area, dM, are (respectively)

$$(10) \quad dX^A/dM = -rcK_M/[Dp(h^A)] = -rK_M qX^A/D > 0;$$

$$(11) \quad dE^A/dM = [rK_M(1+\varepsilon)qE^A]/D \gtreqless 0 \text{ if } |\varepsilon| \gtreqless 1.$$

As shown in Figure 1, with an elastic demand, the effect of mangrove deforestation is to cause X^A and E^A to fall.

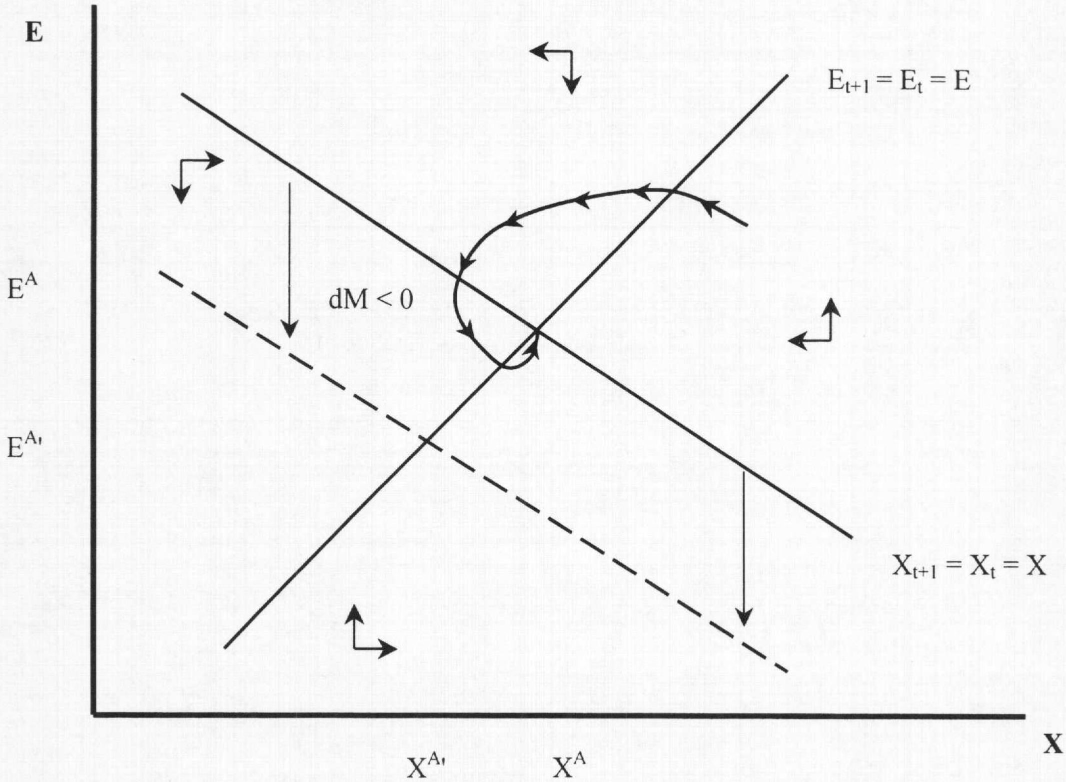
Substituting (5) and (6) into the Schaefer harvesting function $h = qXE$ also yields the long-run supply curve of the fishery (for any given price, $p[h^A]$):

$$(12) \quad h^A = qX^A E^A = rc/[p(h^A)q][K(M) - c/[p(h^A)q]].$$

It is straightforward to demonstrate that this equilibrium supply curve has the normal backward-bending shape for an open-access fishery (Clark, 1976). From (12), it is clear that a change in mangrove area will shift the fishery's long-run supply curve. However, the resulting impact of the supply shift on the equilibrium harvest of fish sold will depend on the elasticity of demand faced by the open access fishery.[4] Using (10) and (11) and

4. In the special case of an infinitely elastic demand, price is unaffected by harvest levels, and in Figure 1 the $E_{t+1} = E_t = E$ isocline defined by zero equilibrium rents will be vertical in $X - E$ space. As a result, a change in mangrove area will affect equilibrium effort levels only, and the corresponding change in open-access equilibrium harvest will be equivalent to the full shift in the fishery's supply curve as a result of a change in mangrove area. See Barbier and Strand (1998), who calculate the marginal impact of a change in mangrove area on harvest levels in an equilibrium open access fishery facing an infinitely elastic market demand (i.e., $p(h) = p$).

FIGURE 1
Mangrove Loss and Long-Run Equilibrium of an Open-Access Fishery, $|\varepsilon| > 1$



assuming that the mangrove-carrying capacity effect in the biological growth function takes the form $K(M) = \alpha \ln M$, $K_M = \alpha/M > 0$, then the marginal impact of a change in mangrove area on equilibrium harvest is

$$(13)\ dh^A/dM = qX^A(dE^A/dM) + qE^A(dX^A/dM) = \varepsilon h^A rqK_M/D = \varepsilon h^A rq\alpha/MD > 0 \quad \text{if } D < 0.$$

Because demand has a finite elasticity, one can also determine the consumer surplus impacts of a change in mangrove area.[5] This is easily demonstrated in the isoelastic case. For example, let $p(h) = kh^{\eta}$, $\eta = 1/\varepsilon < 0$. It follows that the change in consumer surplus, ΔS, resulting from a change in equilibrium harvest levels (from h^0 to h^1) is

$$(14)\ \Delta S = \int_{h^0}^{h^1} p(h)\,dh - [p^1h^1 - p^0h^0] = [k\lfloor(h^1)^{\eta+1} - (h^0)^{\eta+1}\rfloor]/(\eta+1) - k[(h^1)^{\eta+1} - (h^0)^{\eta+1}] = -(\eta[p^1h^1 - p^0h^0])/(\eta+1).$$

By utilizing (13) and (14) it is possible to estimate the new equilibrium harvest and price levels and thus the corresponding changes in consumer surplus associated with mangrove deforestation, for a given demand elasticity, ε.[6]

5. Because rents are dissipated in the open access equilibrium, producer surplus is zero. The welfare impacts due to a change in mangrove area affect consumer surplus only. See Barbier (2000) and Freeman (1991) for further discussion.

6. The following analysis estimates the impacts of mangrove deforestation on the stable equilibrium (X^A, E^A) of the open-access fishery, assuming that effort and stocks adjust instantaneously to allow a new equilibrium to be attained. The author does not consider the case where the effect of deforestation is to make a steady-state equilibrium infeasible, thus causing the

Note that condition (13) is depicted in terms of equilibrium harvest, h^A, mangrove area, M, and effort, E^A, the bioeconomic parameters (α, r, and q) of the model, and the equilibrium price and cost parameters (ε p^A, and c). All of these variables and parameters are observable for Thailand's fisheries, except for the bioeconomic parameters. Following the same approach as Barbier and Strand (1998), it is possible to derive approximations of the latter unknowns. This approach essentially involves employing equilibrium condition (5) to establish a relation between shrimp production, effort and mangrove area that can be estimated with empirical data. For example, by substituting for $X = h/qE$ in condition (5) and rearranging, one obtains

$$(15) \qquad h = qEK(M) - (q^2/r)E^2$$
$$= q\alpha E \ln M - (q^2/r)E^2.$$

Estimation of equation (15) will therefore yield the parameters $b_1 = \alpha q$ and $b_2 = -q^2/r$. To estimate this regression, data are pooled on the artisanal shellfish and demersal fisheries, as well as on mangrove area, across the five coastal zones of the Gulf of Thailand and the Andaman Sea and over the time period 1983–96. Separate regressions are conducted for the shellfish and demersal fisheries. For the respective fisheries in each zone, h is represented by annual harvest in kg and E by the total number of hours (in 10^3 units) spent each year on fishing. The latter is estimated by multiplying the number of fishing instruments recorded per annum times the average number of hours spent per fishing instrument per year. As fishing is predominantly artisanal, and the main type of gear is gill nets, the calculation of effort for the respective fisheries in each zone is in terms of the total number of hours spent on fishing by these instruments annually (Sathirathai, 1998). Finally, M is measured in terms the annual mangrove area in km^2 for each zone.

Equation (15) was estimated by utilizing and comparing various regression procedures for a pooled data set: (1) ordinary least squares (OLS); (2) one- and two-way panel analysis of fixed and random effects; (3) a random coefficients approach; and (4) a maximum likelihood estimation by an iterated generalized least squares (GLS) procedure for a pooled time-series–cross-sectional regression, which allows for correction of any groupwise heteroscedasticity, cross-group correlation, and common or within-group autocorrelation. Tables 1 and 2 indicate the best regression models for the demersal and shellfish fisheries respectively, across the five zones and for the 1983–96 period.

For the demersal fishery, the OLS one-way model of the pooled regression is preferred to either the random effects, fixed effects, or the random coefficients panel regressions, but not to the GLS estimations corrected for groupwise heteroscedasticity and cross-group correlation. Table 1 reports the OLS regression and the two preferred GLS regressions. The test statistics indicate rejection of the null of hypothesis of groupwise homoscedasticity and no cross-group correlation but permit the assumption of nonautocorrelated disturbances.[7] Thus the

fishery to switch to a different exploitation path that leads to collapse of the fishery. Although there is evidence that mangrove conversion in southern Thailand is affecting the coastal and marine fisheries, it is unlikely that they are currently in danger of such a near collapse scenario.

7. For the panel analysis of the demersal fisheries, two likelihood ratio tests were performed, a chi-squared and an F-test, to verify the null hypothesis of zero one-way and two-way individual effects across all 5 zones and 14 time periods. Only the chi-squared test for one-way effects was significant. In addition, the Breusch-Pagan Lagrange multiplier (LM) statistic was also highly insignificant for both the one-way and two-models, which suggests that the null hypothesis of zero random disturbances cannot be rejected. Although the chi-squared test of the null hypothesis of no randomness of the coefficients was significant, the t-tests on the individual parameters estimated by the random coefficients model were insignificant, suggesting the null hypotheses $b_1 = 0$ and $b_2 = 0$ cannot be rejected. These tests imply that, in the panel analysis, the OLS regression is likely to be more efficient than either the fixed effects, random effects or random coefficients models. However, as indicated in Table 1, from the pooled time-series–cross-sectional GLS regression, the likelihood ratio (LR) test statistic of the null hypothesis for homoscedasticity based on the least squares regression was computed to be 36.74, which is statistically significant at the 1% level. Although not shown in the table, the alternative Wald test for homoscedasticity is also statistically significant and confirms rejection of the hypothesis. Thus the GLS model with correction of groupwise heteroscedasticity is preferred to the OLS regression. The LM statistic of 19.47 reported in Table 1 is a test for cross-sectional correlation versus homoscedasticity, which proves to be statistically significant. Although not indicated in the table, the LR test statistic for cross-group correlation

TABLE 1
Pooled Time-Series–Cross-Sectional Analysis for Demersal Fisheries, Thailand

Coefficient	OLS	Groupwise Heteroscedastic GLS	Groupwise Heteroscedastic and Correlated GLS[a]
b_0	1,084,103.78** (9.470)	954,621.84** (12.598)	830,423.39** (13.165)
b_1	103.5507** (6.687)	105.8432** (6.369)	125.413** (12.264)
b_2	−0.1676** (−2.834)	−0.1363** (−2.836)	−0.1584** (−3.972)
Log-likelihood	−1,017.35	−998.977	−984.845
No. of observations = 70		Likelihood ratio statistic = 36.74**	
F-test = 10.59**		Lagrange multiplier statistic = 19.47*	
		Autocorrelation coefficient (r) = 0.483	

Notes: $HD_{it} = b_0 + b_1 EDLM_{it} + b_2 ED2_{it} + \mu_{it}$, where $i = 1, \ldots, 5$ zones, $t = 1, \ldots, 14$ years (1983–96); HD_{it} = demersal fish harvest (kg) for zone i at time t; $EDLM_{it}$ = demersal fisheries effort (hours) × log mangrove area (km^2) for zone i at time t; and $ED2_{it}$ = demersal fisheries effort (hours) squared for zone i at time t. t-statistics in parentheses.
[a]Preferred model.
*Significant at 95% confidence interval.
**Significant at 99% confidence interval.

preferred model for demersal fisheries shown in Table 1 is the GLS estimation allowing for groupwise heteroscedasticity and cross-group correlation. In the case of shellfish, the one-way random effects model is preferred to all other panel specifications, except for the random coefficients model. In the GLS models, the null hypothesis of groupwise homoscedasticity was also rejected, but in addition significant autocorrelation across zones was found. Table 2 indicates the one-way random effects estimation, the random coefficients model, and the GLS regression allowing for both groupwise heteroscedasticity and common autocorrelation but not for cross-sectionally correlated disturbances.[8]

versus groupwise heteroscedasticity was estimated to be 28.27, which is also statistically significant. Thus the null hypothesis of zero cross-group correlation in the demersal fisheries regression can be rejected. As Table 1 indicates, the common autocorrelation coefficient across all five zones was estimated to be 0.483, and although not shown in the table, the group-specific residual autocorrelation coefficients for each of the five zones respectively were $\rho_1 = 0.968$, $\rho_2 = 0.602$, $\rho_3 = 0.266$, $\rho_4 = 0.067$, and $\rho_5 = 0.515$. The chi-squared test for the latter coefficients suggests that autocorrelation is significant at the 95% critical level in three of the five zones. In addition, further reestimations of the GLS model that allowed for correction of common or group-specific autocorrelation led to the null hypothesis that the coefficient $b_2 = 0$ no longer being rejected. Thus the preferred GLS model involves correction for groupwise heteroscedasticity and cross-group correlation, but not for common or within-group autocorrelation.

8. In all regressions of (15) for shellfish, inclusion of a constant term led to significant distortions in the estimations of the parameters b_1 and b_2. The constant term was therefore omitted. For the panel analysis of shellfish two LR tests were performed, a chi-squared and an F-test, to verify the null hypothesis of zero individual one-way and two effects. The tests suggest rejection of the two-way specification. However, both tests were highly significant with respect to one-way effects, indicating that pooled OLS is not an efficient panel estimator. The Breusch-Pagan LM statistic of 157.73 reported in Table 2 is also highly significant, which suggests rejection of the null hypothesis of zero random disturbances. In Table 2, the Hausman test statistic is 5.76, which is insignificant and indicates that the null hypothesis of random individual effects across the provinces cannot be rejected. Thus the panel analysis test statistics argue in favor of the random effects model over the fixed effects and OLS regressions. However, in the one-way random effects specification, the coefficient for b_2 is statistically insignificant, whereas both estimated coefficients are statistically significant in the GLS random coefficients model. As reported in Table 2, the chi-squared test of the null hypothesis of no randomness of the coefficients was also highly significant. The random coefficients model is therefore the preferred panel specification. In the case of the pooled time-series–cross-section GLS models, there were significant problems of autocorrelation. The common autocorrelation coefficient across all five zones was estimated to be 0.664, and although not shown in Table 2, the group-specific residual autocorrelation coefficients for each of the five zones respectively were $\rho_1 = 0.814$, $\rho_2 = 0.642$, $\rho_3 = 0.424$, $\rho_4 = 0.789$,

TABLE 2
Pooled Time-Series–Cross-Sectional Analysis for Shellfish, Thailand

Coefficient	One-Way Random Effects	Random Coefficients GLS[a]	Groupwise Heteroscedastic and Autocorrelated GLS
b_1	99.7874 (1.606)	481.8555** (2.730)	123.0386** (6.597)
b_2	−0.0318 (−0.703)	−0.4893* (−1.974)	−0.448 (−1.776)
Log-likelihood	−1,048.761	−998.977	−1,045.1074
No. of observations = 70 Breusch-Pagan (LM) test = 157.73** Hausman test = 5.76		Homogeneity test= 214.97**	Likelihood ratio statistic = 63.41** Lagrange multiplier statistic = 8.557 Autocorrelation coefficient (r) = 0.664

Notes: $HS_{it} = b_1 ESLM_{it} + b_2 ES2_{it} + \mu_{it}$, where $i = 1, \ldots, 5$ zones, $t = 1, \ldots, 14$ years (1983–96); HS_{it} = shellfish harvest (kg) for zone i at time t; $ESLM_{it}$ = shellfish fisheries effort (hours) × log mangrove area (km^2) for zone i at time t; $ES2_{it}$ = shellfish fisheries effort (hours) squared for zone i at time t. t-statistics in parentheses.
[a]Preferred model.
*Significant at 95% confidence interval.
**Significant at 99% confidence interval.

The preferred estimation for shellfish is the random coefficients model.

As indicated in Tables 1 and 2, the estimated coefficients in the preferred regressions for both the shellfish and demersal fisheries are highly significant. Although the two coefficient estimates on their own are insufficient to determine explicitly the three bioeconomic parameters (α, r, and q) for the two artisanal fisheries, from (15) $b_1 = \alpha q$ and $b_2 = -q^2/r$. It follows that condition (13) can now be rewritten as

$$(16)\quad dh^A = [(\varepsilon h^A b_1)/(M[-b_2(1+\varepsilon)E^A - [c/p(h^A)]])]\, dM.$$

and $\rho_5 = 0.650$. The chi-squared test for the latter coefficients suggests that autocorrelation is significant at the 95% critical level in all five zones. Although GLS models that corrected for both common and within-group correlation were estimated, the former models performed best. The LR statistic of 63.41 reported in Table 2 for the autocorrelated-corrected GLS regression is statistically significant. Although not shown in the table, the alternative Wald test for groupwise homoscedasticity is also statistically significant and confirms rejection of the null hypothesis. However, the LM statistic of 8.55 reported in Table 2 is not statistically significant, and the LR test statistic for cross-group correlation versus groupwise heteroscedasticity was estimated to be 13.33, which is also not statistically significant. Thus the null hypothesis of zero cross-group correlation in the shellfish regression cannot be rejected. Nevertheless, because the estimated coefficient for b_2 is not statistically significant for the GLS regression allowing for groupwise heteroscedasticity and common autocorrelation, the preferred estimation of (15) for shellfish is the GLS random coefficients model.

Once the effects of a change in mangrove area on equilibrium harvest are imputed through employing (16), it is therefore possible to use (14) to estimate the corresponding changes in welfare for the shellfish and demersal fisheries. In Section V, these welfare effects are estimated explicitly for the base case simulation in which mangrove deforestation occurs in Thailand at the annual average rate of 5,183 ha over 1983–96. However, before discussing this base case scenario and the simulation scenarios for macroeconomic impacts, it is necessary to develop the second model of the economic factors influencing conversion of mangroves to shrimp ponds in Thailand.

IV. A MODEL OF MANGROVE CONVERSION FOR SHRIMP FARMING

If shrimp farm expansion is a major cause of mangrove loss, then the resulting deforestation rate in any time period, r_t, must be related to the amount of mangrove area cleared by shrimp farms, that is,

$$(17)\qquad M_{t+1} - M_t = -r_t = N_t,$$

where the left-hand side is the rate of change in mangrove area, M, over time and N_t is the amount of land cleared for shrimp farming. The latter depends in turn on the factors influencing the mangrove clearing decisions of shrimp farmers.

In any time period, t, let the profit function of a representative shrimp farm be the profits gained by choosing a profit-maximizing level of inputs

$$(18) \quad \max \pi(p, w, w_N) = \max_{N_j, x_j} pf(x_j, N_j) - wx_j - w_N N_j,$$

where the variable inputs include the amount of mangrove area cleared by the jth farm, N_j, and a vector, x_j, of other $i, \ldots, k$ inputs (e.g., labor and feed/fertilizer) used in production of shrimp. The corresponding vector of input prices is w, and p is the price of shrimp. Finally, w_N is the rental price of land. If shrimp farmers clear their own land from mangroves, then this is an implicit price or opportunity cost (Panayotou and Sungsuwan, 1994). However, if the farm purchases or rents additional cleared land from a market, then w_N would be the market rental price of cleared mangrove land (Cropper et al., 1999).

Utilizing Hotelling's lemma, the derived demand for cleared land by the jth shrimp farm, N_j, is therefore

$$(19) \quad N_j = N_j(p, w, w_N) = -(\partial\pi/\partial w_N), (\partial N_j/\partial w_N) < 0, \quad (\partial N_j/\partial p) > 0.$$

Equation (19) depicts the derived demand for cleared mangrove area by the representative jth farm. In aggregating this demand for cleared land across all j shrimp farms in a province or coastal zone, it is important to consider other factors that may influence the aggregate level of conversion, such as income per capita, population, land area, and the aggregate number of farms. Thus, allowing Z to represent one or more of these exogenous factors and N the aggregate demand for cleared mangrove area in a province or coastal zone, the latter can be specified as

$$(20) \quad N = N(p, w, w_N; Z).$$

As shrimp farms generally provide their own supply of cleared mangrove land, N, one can view this type of own supply as a kind of production of cleared land, which is governed by the following conditions. The source of the cleared land (i.e., a mangrove swamp) is an open-access resource, so that land is cleared up to the point where any producer surpluses (rents) generated by clearing additional land are zero. The principal input into clearing land is labor, L, which is paid some exogenously determined wage rate, w_L, and the production function is assumed to be homogeneous. This production of cleared land may also be affected by a range of exogenous factors, α, that may influence the accessibility of pristine mangrove areas available for conversion, including roads, infrastructure, and proximity to major cities and towns.

Thus one can specify a cost function, based on the minimum cost for the shrimp farm of producing a given level of cleared mangrove area, N, for some fixed levels of w_L and α, as

$$(21) \quad C_j = C_j(w_L, N; \alpha).$$

Under open-access conditions, each shrimp farm will convert mangrove area up to the point where the total revenues gained from converting N_j units of land, $w_N N_j$, equal the total costs represented by (21). If a shrimp farm clears its own mangrove area, then w_N is now the household's implicit rental price, or opportunity cost, of utilizing additional converted land. However, as the shrimp farm is essentially supplying land to itself, then in equilibrium the implicit price ensures that the farm's costs of supplying its own land will be equated with its derived demand for converted mangrove area. Then for the jth representative household the following cost conditions for supplying its own cleared land must hold

$$(22) \quad w_N = c_j(w_L, N_j; \alpha), \quad (\partial c_j/\partial w_L) > 0, (\partial c_j/\partial N_j) > 0, \quad (\partial c_j/\partial \alpha) < 0, j = 1, \ldots, J.$$

The right-hand side of (22) is the average cost curve for clearing land, which may be increasing with the amount of land cleared as, among other reasons, one must venture further into the mangrove swamp to clear more land (Angelsen, 1999). That is, in equilibrium, the shrimp farm's implicit price for cleared land will be equated with its per unit costs of mangrove conversion. Together with the farm's derived demand for converted land (19), equation (22) determines the equilibrium level of mangrove clearing by the shrimp farm as well as its implicit price.

Although the latter variable is not observed, it is possible to use (19) and (22) to solve for the reduced-form equation for the equilibrium level of cleared mangrove area. Substituting (22) for w_N in equation (19), and then rearranging to solve for N_j yields

$$(23) \qquad N_j = N_j(p, w, w_N[w_L, \alpha]),$$
$$(dN_j/dw_L) = (\partial N_j/\partial w_L) + (\partial N_j/\partial w_N)(\partial w_N/\partial w_L),$$
$$(dN_j/d\alpha) = (\partial N_j/\partial w_N)(\partial w_N/\partial \alpha) > 0.$$

Note that the overall impact of a change in the wage rate on mangrove clearing by a shrimp farmer is ambiguous, due to two possible counteracting effects. Given equation (22), one would expect that a higher wage would make it more costly for the farmer to clear mangrove area, thus reducing the equilibrium amount of land converted (i.e., $[\partial N_j/\partial w_N][\partial w_N/\partial w_L] < 0$). However, because labor is also used in shrimp production, equation (19) indicates that the wage rate is included in the vector of input prices, w, that influence the demand for cleared land. If land and labor are substitutes in shrimp farm production, then one would also expect a rise in the wage rate to increase the use of converted mangrove land in aquaculture (i.e., $[\partial N_j/\partial w_L] > 0$). Thus whether the equilibrium level of cleared land will increase or decrease in response to a rise in the wage rate will depend on the relative magnitude of these two effects.

Aggregating (23) across all J shrimp farms in a province or coastal zone that convert their own land from mangrove areas, and including exogenous factors Z, leads to a reduced-form relationship for the aggregate equilibrium level of cleared mangrove land

$$(24) \qquad N^* = N(p, w_I, w_L; \alpha, Z),$$
$$(dN/dp) > 0, (dN/d\alpha) > 0,$$

where the wage rate, w_L, is now distinguished from the vector of prices for inputs other than labor, w_I. The amount of mangrove land converted should increase with the price of shrimp and accessibility of the mangrove forest. However, as in the case of (23), the impact of a change in the wage rate or other input prices is ambiguous. Returning to (17), it follows that

$$(25) \qquad M_{t+1} - M_t = -r_t = N_t(p, w_I, w_L; \alpha, Z)$$

Thus the mangrove deforestation relationship (25) can be empirically estimated, using appropriate data for the shrimp output price, p, the wage rate, w_L, other input prices, w_I, the "accessibility" of mangrove areas, α and other economic factors that may affect the mangrove clearing decision, Z. The regression was estimated by employing pooled data across the five coastal zones of the Gulf of Thailand and the Andaman Sea and over the time period 1983–96. The shrimp price variable was approximated by the export unit value of shrimp converted to Thai currency, which is common to all five zones. The real minimum wage rate for each zone was employed as the proxy for w_L. Although the real ammonium phosphate price was used as the proxy for w_I, this variable proved not to be significant in the regression, and its inclusion also affected the significance of the coefficients of other variables. Ammonium phosphate price was therefore dropped from the estimation. The accessibility of mangrove areas, α, was represented by the distance of each zone from the Thai capital, Bangkok. Finally, the exogenous economic factors, Z, included the shrimp farm density of each zone, the real per capita gross product in each zone and the real rate of interest, which is common to all zones.[9]

Equation (25) was estimated through various regression procedures for a pooled data set: OLS; one- and two-way panel analysis of fixed and random effects; and a maximum likelihood estimation by an iterated GLS procedure.[10] The preferred regression is the GLS model correcting for groupwise heteroscedasticity, cross-group correlation, and group-specific autocorrelation.[11] Table 3 reports the parameter estimates and elasticities resulting from

9. Additional exogenous factors that were also tried in the regressions of (25) were the population density of each zone, shrimp farm yields per zone, and average shrimp farm size per zone. However, none of these variables were significant in the regressions, and their inclusion affected the coefficients of other variables.

10. Due to the limited degrees of freedom, the random coefficients approach could not be estimated.

11. For the panel analysis of mangrove deforestation two LR tests were performed, a chi-squared and an F-test, to verify the null hypothesis of zero individual

TABLE 3
Pooled Time Series–Cross-Sectional Analysis of Mangrove Deforestation, Thailand

Variables	Sample Means	Parameter Estimates	Elasticity Estimates
P_{it}	193.07	0.0442(2.149)*	0.8240
EV_{it}	7.64	1.1187(2.149)*	0.8249
ER_{it}	25.29	0.3381(2.149)*	0.8249
w_{it}	79.07	0.2756(2.643)**	2.1724
FD_{it}	0.13	−35.0457(−2.307)**	−0.4538
D_{it}	566.55	−0.0327(−4.678)**	−1.7897
R_{it}	12.12	0.0891(0.543)	0.1042
Y_{it}	42,305.46	−0.00035(−2.551)**	−1.4194
No of observations = 60		Likelihood ratio statistic = 74.53**	
Autocorrelation coefficient (r) = 0.7299		Lagrange multiplier statistic = 34.07**	
Group-specific autocorrelation coefficients ($\rho_1 = 0.613, \rho_2 = 0.728, \rho_3 = 0.871,$ $\rho_4 = 0.929$, and $\Delta_5 = 0.509$)			

Notes: $M_{it} - M_{it+1} = \beta_0 + \beta_1 P_{it} + \beta_2 w_{it} + \beta_3 FD_{it} + \beta_4 D_{it} + \beta_5 R_{it} + \beta_6 Y_{it} + \mu_{it}$, where $i = 1, \ldots, 5$ zones, $t = 1, \ldots, 14$ years (1983–96); $M_{it} - M_{it+1}$ = change in mangrove area (km^2) from t to $t+1$, for zone i at time t; $P_{it} = EV_{it} * ER_{it}$, where EV_{it} is the export unit value of shrimp (US\$/kg) and ER_{it} is the exchange rate (baht/US\$), for all zones at time t; w_{it} = minimum real wage rate (baht/day), for zone i at time t (1990 prices); FD_{it} = number of shrimp farms per total land area (km^2), for zone i at time t; D_{it} = distance (km) of each zone i from Bangkok, for all time t; R_{it} = real rate of interest (%), for all zones at time t (1990 prices); Y_{it} = real per capita gross product (baht), for zone i at time t (1990 prices). Mean of dependent variable ($M_{it} - M_{it+1}$) = 10.3652. t-statistics in parentheses.
*Significant at 95% confidence interval.
**Significant at 99% confidence interval.

the latter regression. All the explanatory variables appear to be significant in affecting mangrove loss except for real interest rates.

The regression results yield the signs predicted by the model of mangrove clearing for shrimp farming (see equation [24]). An increase in the price of shrimp leads to greater mangrove deforestation, and more accessible mangrove areas (i.e., those closer to Bangkok) are also subject to increased deforestation. However, a rise in the real wage has an unambiguously positive impact on mangrove loss. As the discussion following equation (23) suggests, although the amount of mangrove land converted should decrease with the cost of labor, which is the principal input involved in clearing operations, this effect may be counteracted by an opposing impact of a rise in the wage rate on mangrove

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one-way and two effects. The tests suggest rejection of the two-way specification, and only the chi-squared test for one-way effects was significant. In addition, the Breusch-Pagan LM statistic was also highly insignificant for both the one-way and two-models, which suggests that the null hypothesis of zero random disturbances cannot be rejected. These tests imply that in the panel analysis, the OLS regression is likely to be more efficient than either the fixed effects or random effects models. However, as indicated in Table 3, from the pooled time series-cross sectional GLS regression, the LR test statistic of the null hypothesis for homoscedasticity based on the least squares regression was computed to be 74.53, which is statistically significant at the 1% level. Although not shown in the table, the alternative Wald test for homoscedasticity is also statistically significant and confirms rejection of the hypothesis. Thus the GLS model with correction of groupwise heteroscedasticity is preferred to the OLS regression. The LM statistic of 34.07 reported in Table 3 is a test for cross-sectional correlation versus homoscedasticity, which proves to be statistically significant. Although not indicated in the table, the LR test statistic for cross-group correlation versus groupwise heteroscedasticity was also statistically significant. Thus the null hypothesis of zero cross-group correlation in the demersal fisheries regression can be rejected. As Table 3 indicates, the group-specific residual autocorrelation coefficients for each of the five zones respectively were $\rho_1 = 0.613$, $\rho_2 = 0.728$, $\rho_3 = 0.871$, $\rho_4 = 0.929$, and $\rho_5 = 0.509$. The chi-squared test for the latter coefficients suggests that autocorrelation is significant at the 95% critical level in all five zones. In addition, further reestimations of the GLS model that allowed for correction of group-specific autocorrelation led to improved significance in the coefficients of the explanatory variables of the regression. Thus the preferred GLS model involves correction for groupwise heteroscedasticity, cross-group correlation, and within-group autocorrelation.

conversion, if land and labor are substitutes in shrimp farming. The positive estimated coefficient for w suggests that this latter substitution effect might be the stronger influence. As the costs of labor use in production rise, shrimp farmers may be induced to move from more intensive aquaculture operations that employ relatively more labor than land to more semi-intensive and extensive systems that require relatively more land.[12]

The regression results in Table 3 also indicate that an increase in shrimp farm density is associated with lower mangrove deforestation. An increase in the number of shrimp farms in a given zonal area may be an indication that more intensive shrimp farming operations are being adopted, and because such operations require less land, overall clearing of mangrove areas is lower. Alternatively, greater shrimp farm density may also imply that there is less remaining mangrove area available to convert, and as a consequence, deforestation will slow down. A rise in the real per capita income in a zone appears also to reduce mangrove deforestation. One possible explanation is that increases in GDP per capita in a coastal zone is associated with greater economic opportunities other than shrimp farming are occurring in the zone. Thus resources are shifted away from investing in shrimp farm expansion to other investments, and as a result clearing of mangroves is reduced.

The elasticity estimates in Table 3 indicate that real wages have the strongest impact on mangrove loss, followed by the distance of each zone from Bangkok, income per capita, shrimp price, and shrimp farm density. A 10% rise in real wages will cause mangrove loss to increase by nearly 22%, whereas a 10% increase in shrimp prices will lead to an 8.2% increase. The elasticity for D_{it} illustrates the importance of the accessibility of mangrove areas to deforestation. For example, a coastal zone that is located 50% closer to Bangkok (i.e., around 285 km away, rather than the average distance of about 570 km) is likely to have nearly double the amount of mangrove loss. Finally, a 10% increase in income per capita and shrimp farm density in a zone will cause mangrove deforestation to fall by 14% and 4.5%, respectively.

V. WELFARE EFFECTS OF POLICY SIMULATIONS

Together, the estimation of (25) reported in Table 3 along with equations (16) and (14) can be used to simulate the effects of changes in mangrove area on the corresponding changes in welfare for the shellfish and demersal fisheries. These welfare effects are first estimated for a base case simulation in which mangrove deforestation occurs in Thailand at the annual average rate of 5,183 ha over 1983–96. The corresponding losses to the shellfish and demersal fisheries are then compared to the annual value of shrimp aquaculture, which is calculated as total annual production valued at the border-equivalent US$ price in 1996. However, the estimation of equation (25) suggests that the annual loss in mangrove area through clearing by shrimp farmers will be affected by a number of macroeconomic factors, such as the exchange rate, which affects the border-equivalent price of shrimp, and the level of GDP per capita in coastal zones. As a result of the financial crisis that occurred in Thailand in July 1997, the economy went through a series of macroeconomic shocks that led to both a substantial devaluation of the Thai baht and a fall in GDP per capita.[13] The devaluation will have stimulated exports from Thailand, including the returns from shrimp farming, and thus would precipitate more mangrove clearing. Equally, the regression of (25) suggests that falling GDP per capita across coastal zones will also increase mangrove loss in Thailand. Thus the welfare effects of these two macroeconomic consequences on both coastal fisheries and aquaculture production can also be simulated and compared to the base case scenario.

12. There is some anecdotal empirical evidence corroborating this result. For example, Tokrisna (1998) estimates that in Thailand extensive shrimp farms (5–7 ha) have average labor costs of only $13.5/ha, semi-intensive farms (3–4 ha) have labor costs of $239/ha, and intensive farms (2–3 ha) have labor costs of $922/ha. Thus, a rise in wages may lead some shrimp farmers to expand shrimp farm area and switch to less intensive operations to save on overall labor costs.

13. According to the Asian Development Bank (2000), real GDP per capita (1988 prices) fell by 1.6% in Thailand between 1996 and 1997 and by 10% between 1997 and 1998. The population of Thailand was rising by 1% over these two years. The baht per US$ exchange rate (average of period) rose by 23.8% between 1996 and 1997 and by 32% between 1997 and 1998. The simulations used in this article, a 20% devaluation and a 5% fall in GDP per capita, are therefore representative of the post–July 1997 economic crisis period in Thailand.

TABLE 4
Simulations of Welfare Impacts of Devaluation and GDP per Capita Decline

Scenarios	Annual Loss in Mangrove Area (ha)	Annual Welfare Loss (US$) in Demersal Fish	Shellfish	All Fish
Base case	5,183	62,474	1,251,086	1,313,560
1. 20% devaluation	6,038	72,805	1,462,214	1,535,019
(% change over base case)	(16.50)	(16.54)	(16.88)	(16.86)
2. 5% fall in GDP per capita	5,550	66,917	1,341,742	1,408,659
(% change over base case)	(7.10)	(7.11)	(7.25)	(7.24)
3. Both effects	6,405	77,251	1,553,461	1,630.712
(% change over base case)	(23.59)	(23.65)	(24.17)	(24.14)

Notes: In all scenarios, the demand elasticity for fish is assumed to be −0.5. Base case scenario calibrated to 1996 for all variables except mangrove deforestation (1983–96 average). Annual border-equivalent value of shrimp aquaculture production in 1996 was $1,427 million.

Table 4 summarizes the results of the base case and scenario simulations. The base case is calibrated for pre-economic crisis conditions and, as noted, assumes that mangrove deforestation continues at the average annual rate occurring over 1983–96 of 5,183 ha. Three additional scenario simulations are calculated to reflect the impact of the economic crisis in Thailand: (1) a 20% devaluation in the Thai baht with respect to the US $; (2) a 5% fall in GDP per capita on average across all coastal zones; and (3) the combined effects of both macroeconomic impacts.

The base case scenario depicted in Table 4 illustrates several important aspects about the process of mangrove deforestation in Thailand before the 1997 economic crisis. First, the decline in mangrove forests of around 5,000 ha per year on average did generate substantial welfare impacts on the artisanal coastal fisheries. These economic losses amounted to $1.31 million per year, or about $253/ha of deforested mangrove area. Second, by far the largest impacts occurred in the market for shellfish. Whereas losses to the dermersal fisheries were around $62,500 per year, shellfish losses were about $1.25 million per year. Finally, it is also clear from the base case scenario why throughout the 1980s and 1990s the Thai government chose largely to ignore the welfare impacts on coastal fisheries due to mangrove deforestation from shrimp farm expansion. Although large in absolute terms, these fishery losses are relatively insignificant compared to the value of shrimp aquaculture production in Thailand. In 1996, the value of this production in Thailand was around $1.43 billion. This implies that the welfare impacts on coastal fisheries of mangrove deforestation amounted to less than 0.1% of the border-equivalent value of shrimp production. As noted in the introduction, it is not surprising that the main policy emphasis in coastal areas was to expand coastal aquaculture production to increase export earnings from shrimp production, despite the economic consequences of the accompanying deforestation for coastal fisheries.

As the first simulation depicted in Table 4 indicates, a devaluation of 20% will lead to more clearing of mangroves. Annual mangrove deforestation increases to just over 6,000 ha, and total fishery losses rise by almost 17% to around $1.54 million. However, the devaluation will also increase shrimp export earnings and should also cause expansion in shrimp aquaculture production. For example, ignoring the latter effects, the price effects alone of the devaluation would have increased the value of shrimp aquaculture production in 1996 to $1.71 billion. Compared to this increase, the additional fishery losses due to greater mangrove clearing are clearly negligible.

The second simulation indicates that a fall in per capita GDP of 5% would also lead to greater mangrove clearing and fishery losses. Annual mangrove deforestation increases to over 5,500 ha, and the welfare impacts on the fisheries rise by over 7% to around $1.41 million per year. Although it is unlikely that the GDP decline would affect the border-equivalent price of shrimp in Thailand, it is possible that shrimp aquaculture production might increase under this scenario. As noted in the previous section, one reason why a fall in income per capita leads to greater mangrove clearing is that economic resources would be switched from other investments to shrimp farm expansion. If this is the case, then this income effect should result in greater overall aquacultural production as well.

Finally, the full effect of the 1997 economic crisis in Thailand involved both a substantial devaluation and a sharp fall in GDP per capita. The third scenario in Table 4 shows the impacts of the combined effects of the crisis. In this scenario, mangrove clearing rises to 6,400 ha annually, and welfare losses in coastal fisheries increase over 24% to $1.63 million. However, the devaluation and the fall in GDP per capita are also likely to lead to an increase in shrimp aquaculture production and export earnings. As noted, the price effects of the devaluation alone would increase the border-equivalent value of Thailand's shrimp aquaculture production in 1996 to around $1.71 billion.

VI. CONCLUSION

The rapid increase in mangrove deforestation globally has raised concern by both ecologists and economists that loss of this critical habitat is having a detrimental impact on many coastal fisheries. This concern is particularly acute with respect to Thailand, because recent decades have seen the devastation of mangrove systems in southern Thailand as a result of the rapid expansion of shrimp aquaculture production in coastal areas. Although as a consequence Thailand has become the largest exporter of shrimp internationally and earns $1–$2 billion annually from these exports, the potential impacts on coastal artisanal fisheries have been largely ignored. The welfare effects on coastal fisheries of mangrove conversion to shrimp aquaculture have never been estimated, either in Thailand or elsewhere in the world.

To shed some light on this problem, the approach of this article has been to analyze simultaneously the impacts of shrimp farming on mangrove systems as well as the corresponding effects of mangrove loss on habitat–fishery linkages. The latter linkages were investigated by developing a bioeconomic model of an open-access fishery that depends on coastal mangrove habitat for breeding and nursery grounds. By estimating the key parameters of the model through pooled time-series and cross-sectional data over the 1983–96 period for the five coastal zones of Thailand, it was possible to determine explicitly how mangrove loss affects the economic value of Thailand's coastal fisheries. In addition, the article developed a model of the mangrove clearing decisions of shrimp farmers under open-access conditions. The resulting aggregate reduced-form level of mangrove clearing by all farmers in coastal areas was expected to be a function of the returns and input costs to shrimp farming, plus exogenous economic factors, and this relationship was also empirically estimated across the five coastal zones in Thailand over 1983–96. Finally, both the habitat–fishery model and the mangrove deforestation model are linked to estimate the effects of mangrove loss in terms of the welfare impacts on Thailand's coastal fisheries. These welfare losses were then compared to the annual economic value of shrimp production in Thailand, as well as to simulations to determine the welfare impacts of the recent economic crisis in Thailand.

The most striking result to emerge from the analysis is that it demonstrates why Thailand has chosen to expand coastal aquaculture production to increase export earnings from shrimp production, despite the economic consequences of the accompanying deforestation for coastal fisheries. Although large in absolute terms, the economic losses to coastal fisheries in Thailand from mangrove deforestation are very small relative to the total value of shrimp aquaculture production. Under all simulations, fishery losses ranged from $1.3 to $1.6 million annually. Yet in 1996 the border-equivalent value of shrimp production was over $1.4 billion and under some scenarios could have increased to at least $1.7 billion.

However, this result does not necessarily mean that the economic losses imposed on coastal fisheries from mangrove deforestation should be ignored.

First, local coastal communities in Thailand bear the burden of these losses, yet they share very little of the benefits of shrimp aquaculture production. Much of the financial investment in coastal shrimp farms is from wealthy individual investors and business enterprises from outside of the local community (Flaherty and Karnjanakesorn, 1995). Although some hiring of local labor occurs, in the past shrimp farm owners have tended to hire Burmese workers because their wage rates are much lower.

Second, mangrove forests provide other important values in addition to their role as breeding and nursery grounds for coastal fisheries. Local communities also directly exploit mangroves for subsistence and cash income from harvesting timber, fuel wood, and other wood products, as well as nonwood resources, such as birds and crabs. Other valuable ecological functions of mangroves include carbon sequestration, control of flooding, and protecting the coastline from erosion. Sathirathai and Barbier (2000) have estimated the annual net income from timber and nontimber products to a local community in Surat Thani province to be around $87 per ha of mangrove forest and the coastline protection function of mangroves to be around $3,800/ha. In comparison, this article has estimated the economic losses in habitat–fishery linkages from mangrove deforestation to be $253/ha

Finally, simply because the aggregate production value of shrimp aquaculture is extremely high does not necessarily imply that conversion of mangroves to shrimp ponds is always the economically best use of these coastal areas. As in many areas of the world, shrimp farming in Thailand is heavily subsidized. The de facto open-access availability of mangrove swamps means that the investor incurs only the direct (mainly labor and dredging) costs of conversion and usually pays nominal land rent and taxes (if any) to the government after conversion. A major external cost of shrimp ponds is the considerable amount of water pollution they generate. This consists of both the high salinity content of water released from the ponds and agrochemical runoff. In addition, there is the problem of the highly degraded state of abandoned shrimp ponds after the five-year period of their productive life. Across southern Thailand those areas with abandoned shrimp ponds degenerate rapidly into wasteland because the soil becomes very acidic, compacted, and too poor in quality to be used for any other productive use, such as agriculture. In addition, without considerable additional investment in restoration, these areas do not regenerate into mangrove forests. Finally, many of the conventional inputs used in shrimp pond operations are subsidized, below border-equivalent prices, thus further increasing the private returns to shrimp farming.

Sathirathai and Barbier (2001) conducted an economic cost-benefit analysis of the returns to commercial shrimp farming in southern Thailand, which includes accounting for the external costs of water pollution and rehabilitating the mangrove forest as well as the full economic (i.e., border-equivalent) costs of conventional inputs. The results indicate that excluding the costs of mangrove restoration, the discounted economic returns to commercial shrimp farming range from $194 to $209 per ha. If the costs of regenerating the mangrove forest are included, then the economic returns to shrimp farming are actually negative.[14] In comparison, the net present value of local timber and nontimber harvests from the mangrove forests by a small coastal community is estimated to be around $632 to $823 per ha.[15] As a further comparison, following the estimates from this article suppose that the value of mangrove–fishery linkages in Thailand is $253 per ha annually. Over a 20-year time horizon and assuming discount rates of 10–15%, the net present value of this linkage ranges from

14. All net present value calculations were based on a 20-year time horizon and discount rates of 10–15%. The return to shrimp farm production assumes nondeclining yields over the initial five-year period of investment but assumes no production over years 6–20. Any investment in mangrove restoration is assumed to take place over years 6–20. See Sathirathai and Barbier (2001) for further details.

15. The valuable wood and nontimber products from the mangrove forest collected by local villagers include various fishery products, honey, and wood for fishing poles. The direct use value of these mangrove products was assumed to be equivalent to the net income generated from their harvesting. If the extracted products were sold, market prices were used to calculate the net income generated (gross income minus the cost of extraction). If the products were used only for subsistence, the gross income was estimated based on surrogate prices, that is, the market prices of the closest substitute. These direct use values were estimated to be $88/ha annually, and over the 20-year time period and discount rate assumptions employed for the analysis of shrimp farming, the net present value of the mangrove products ranged from $632 to $823 per ha. See Sathirathai and Barbier (2001) for further details.

$1,587 to $2,158/ha. This is 10 times the discounted economic returns to shrimp farming reported. Clearly, although conversion of mangrove forest into commercial shrimp farming in Thailand is financially attractive to investors, the economic costs of mangrove conversion are extremely high and cannot continue to be ignored by policy makers.

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