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Renewable and Nonrenewable Resource Theory Applied to Coastal Agriculture, Forest, Wetland, and Fishery Linkages

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Abstract *This paper addresses tradeoffs in wetland development using a framework that integrates economic theory of renewable and nonrenewable resources. The theory treats wetland development as use of a nonrenewable resource, while wetland preservation protects critical fishery habitat. The framework recognizes that wetland quality may vary for either development or fisheries. An illustrative application assesses tradeoffs in converting pocosin wetlands to agriculture rather than maintaining wetlands to protect salinity in estuarine nursery areas. Results reveal the marginal value of salinity protection may be substantial, while location may affect a wetland's value to an estuarine shrimp fishery. Comparisons between agricultural and forestry land-uses show that ecological links may cause wetland values to depend upon the land-use chosen for the developed state. Future assessments of other development may reveal additional impacts through impacts on salinity.*

Keywords nonrenewable, renewable, fishery, wetland value, pocosin, Pamlico Sound

Economic development, such as drainage of coastal wetlands, may impose external losses on renewable resource production, such as commercial or recreational fisheries. Several studies address the opportunity costs of preservation (*e.g.*, Brown 1976, Batie and Mabbs-Zeno 1985, Danielson and Leitch 1986, Shabman and Bertelson 1979) and several others address the external costs of development (*e.g.*, Batie and Wilson 1978; Farber 1987; Kahn and Kemp 1985; Lynne *et al.* 1981). Relatively few studies address both preservation and development, perhaps due to the natural division between proponents of each or perhaps due to the difficulty of simultaneously estimating benefits for both. For example, although Stavins (1990) provides a partial exception, most studies (*e.g.*, Gupta and Foster 1975) emphasize the taxonomy of costs and benefits rather than ecological inter-

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dependencies implicit in the wetland allocation issue (*cf.* Crocker and Tschirhart 1992).

This paper addresses tradeoffs implicit in wetland development using a framework suggested by integrating the basic theories of renewable and nonrenewable resource economics (Hotelling 1931; Clark and Munro 1975; Swallow 1990). The model is tailored to assess issues raised by the historical controversy surrounding the drainage of coastal wetlands for agriculture on the Pamlico-Albemarle Peninsula of North Carolina (Heath 1975, NRCD 1987, Street and McClees 1981). Ecologists and fishermen believe that these *freshwater* wetlands, known as pocosin wetlands, may be critical to sustaining commercial fisheries in the Pamlico Sound estuary, particularly the Penaeid shrimp fishery (Street and McClees 1981). This debate arises, in part, because human-induced ecosystem responses may limit economic activity and sustainability (see Crocker and Tschirhart 1992) in the Pamlico fisheries.

Three objectives motivate this paper: 1) to develop a resource-theoretic approach to marginal preservation-development choices; 2) to apply this framework in an assessment of the public issues raised by historical concern regarding pocosin development; 3) to suggest that cognizance of resource interdependencies can clarify policy debates. After providing background on the study area, the paper addresses these objectives in order.

The application relies heavily on the Penaeid shrimp fishery of Pamlico Sound because both biological and economic data are available and because ecological or "natural history" theories are sufficient to suggest a plausible approach to assess biophysical relationships. Furthermore, public concerns often focus on the shrimp fishery because it is by far the largest of Pamlico's fisheries. The assessment-level application offers substantial applied insights as well as illustration, including empirical results showing that wetland values may depend on the alternative land-use under consideration. Such key results are revealed by intermediate steps taken in the empirical assessment of the pocosin development issue.

Study Area: Background and Context

This paper applies a model of interdependent renewable and nonrenewable resources to coastal zone development and the Pamlico Sound, NC, Penaeid shrimp fishery. The background for this application will motivate the theoretical structure for the empirical analysis in subsequent sections.

In the U.S., coastal zone development causes up to 90% of losses in estuarine acreage and indirectly diminishes estuarine productivity through off-site impacts (Tiner 1984). The Pamlico-Albemarle Estuarine Complex is one of the largest and most productive estuarine systems in North America (Epperly and Ross 1986, NRCD 1987). Throughout the U.S., estuarine-dependent fish species comprise 50–90% of commercial landings, with North Carolina landings nearly 90% estuarine-dependent (Epperly and Ross 1986, Street and McClees 1981, Tiner 1984). In Pamlico Sound, Penaeid shrimp support the most prized fishery, producing about 25% of gross dock-side revenues (Purvis and McCoy 1974, Street and McClees 1981). Regionally, however, North Carolina produces a small share (2–4%) of U.S. landings and ex-vessel prices follow the larger Gulf of Mexico or U.S. markets (Waters *et al.* 1980).

Recent public debates concern the conflicts between coastal development and estuarine-dependent vocations (see NRCD 1987). On the preservation side, discussions focus on commercial marine fisheries, especially the highly-valued shrimp fishery (Street and McClees 1981). On the development side, forestry and agriculture dominate the converted freshwater pocosins (peat-bog wetlands) near the estuary (Heath 1975). However, substantial site preparation costs make forestry and agricultural uses profitable marginally, but these uses remain the dominate threat to pocosin wetlands (Heimlich and Langner 1986). No consensus exists concerning what future developers will propose, but predictions range from urbanization to hog and poultry production to peat mining for electricity generation, with public officials anxious to diversify the area's impoverished economy (Richardson 1981, Tiner 1984, NRCD 1987). Agriculture still motivates policy analyses (*i.e.*, Palmquist and Danielson 1989) and therefore appears to provide the highest return.

In this case study, coastal development requires up to 20 miles of drainage canals per square mile (Heath 1975). Drainage of pocosin wetlands irreversibly alters the local hydrologic system by eliminating the vegetative and peat-bog structure that inhibits water flow, causing a decline in the salinity level of estuarine nursery areas (Heath 1975, Skaggs *et al.* 1980, Jones and Sholar 1981, Epperly and Ross 1986). Rehabilitation of the hydrologic function is viewed as impractical since the peat-bog structure may take millennia to regenerate.[1] Thus, wetland development may be assumed irreversible, which is broadly consistent with the ecology of pocosin wetlands and the engineering of their drainage (Heath 1975, NRCD 1987). Irreversibility places wetland development as a nonrenewable resource sector (Krutilla 1967; Fisher and Krutilla 1974).

On the fishery side, Pamlico's juvenile shrimp stock annually arrives via onshore currents; these currents carry juveniles from an open ocean breeding ground to estuarine nurseries along the South Atlantic and Gulf coasts (Epperly and Ross 1986, Williams 1955). Furthermore, if shrimp escape from the Pamlico Sound fishery,[2] they generally will not be harvested elsewhere (Williams 1955, McCoy 1972, Purvis and McCoy 1974, Hettler and Chester 1982, Babcock and Mundy 1985, Matylewich and Mundy 1985). Finally, an altered hydrologic system causes the salinity level to decline in estuarine nurseries, thereby diminishing the survival rate of juvenile shrimp (Jones and Sholar 1981, Street and McClees 1981; Williams 1955). Thus, while a fishery is generally viewed as a renewable resource sector, Pamlico's annual shrimp crop fits the independent-generations fishery model (Wilen 1985). This fishery depends upon environmental factors rather than on a breeding stock.

Economists recognize this process in other fisheries by linking shrimp production to environmental variables (Blomo *et al.* 1982, Griffin *et al.* 1976), but their analyses have not linked estuarine environmental variables directly to development. Irreversible hydrologic impacts suggest wetland development is a type of nonrenewable resource extraction, the key analogy here.

[1] Native Americans used "pocosin" to indicate that these wetlands are perched atop a hill; damming of drainage canals is not expected to completely simulate the sponge-like storage capacity of the original peat soils. Heath (1975) focused on agriculture, while Campbell and Hughes (1991) note that modern forestry practices may still preserve hydrologic functions.
[2] Including Pamlico Sound proper and the Pamlico and Neuse Rivers (U.S. National Marine Fisheries Service areas 6354, 6355, and 7011).

Theory

The theoretical foundation for this analysis derives from well-known theories of exploitation for renewable and nonrenewable resources (Hotelling 1931; Clark and Munro 1975), particularly when these resource sectors are interdependent, such as when the nonrenewable resource provides wetland habitat for the renewable resource (Swallow 1990). With independent generations in the fishery, the shrimp population X is modeled to depend only upon the habitat stocks E:

$$X_t = X(E^H(t), E^N(t)), \tag{1}$$

where E^H and E^N quantify stocks of "high quality" habitat and "normal quality" habitat, respectively; development determines the habitat available at time t. In the renewable sector, fishermen maximize returns or profits R from their labor L and the stock of shrimp, so that renewable resource benefits are:

$$R_t = R(L^*, X(E^H, E^N)) = R^*(X), \tag{2}$$

where L^* is the benefit-maximizing quantity of labor conditional on X, so R^* represents fishery benefits conditional on the shrimp stock X in a given year. In the wetland development sector, nonrenewable resource benefits B^i ($i = H,N$) depend upon the development rate d^i and the available stock of wetlands of a particular type:

$$B^i(d^i, E^i) = C^i(E^i)\, d^i, \; i = H,N, \tag{3}$$

where $C^i(\bullet)$ denotes the net marginal benefit of an additional acre of development for wetland type i, which depends on the acres remaining, E^i. E^i indexes the quality of remaining acres from the perspective of development.

Development of pocosin wetlands requires permits through environmental managers, state officials and the U.S. Corps of Engineers under Section 404 of the U.S. Clean Water Act. Economically, environmental managers must consider the balance between the present value of wetlands preserved for fisheries and the return to wetlands development, at the margin. Then, the economic choice depends on the net opportunity cost (NOC) of wetland development:[3]

$$NOC^i(X, E^H, E^N) = (\partial R^*/\partial X) \cdot (\partial X/\partial E^i) - r \cdot C^i(E^i) \tag{4}$$

where NOC^i is the net opportunity cost of developing wetlands of type i ($i = H,N$). NOC^i measures the loss of fishery profits if a marginal unit of E^i is developed, net of the annualized return to development. If NOC^i is positive, denial of development permits maximizes social returns from wetlands; if NOC^i is negative, then approval of development permits returns marginal development benefits that more than offset losses in the renewable sector. Since wetland quality may be heterogeneous for either preservation or development, NOC may differ for different wetland qualities. The next sections assess the potential for managers to

[3] This marginal condition assumes appropriate concavity of (1)–(3).

find NOC is negative, favoring development, for pocosin wetlands near Pamlico Sound.

Application

The empirical analysis assumes the Pamlico Sound shrimp fishery: 1) is economically and ecologically independent of neighboring shrimp populations; 2) is affected by development; 3) and is a price taker. The link between fishery benefits and coastal development is generated in a stepwise process which 1) estimates the potential fishing rents for a given stock of shrimp, 2) links the shrimp stock to estuarine salinity, and 3) links estuarine salinity to development (Fig. 1). Loomis (1988) used a stepwise model in a forestry-fishery context. Our stepwise approach combines two approaches suggested by Kahn (1987), by including environmental variables in the harvest function while also using an (simple) ecosystem model. This approach permits improvements at any step as new scientific or policy information warrants.

Figure 1 illustrates the basic empirical model. Development initiates changes in potential fishery benefits: drainage of pocosin wetlands alters estuarine salinity which then lowers the expected annual, estuarine production of shrimp; lower estuarine productivity alters the conditions of profit maximization; finally, the shrimp fishery realizes a change in expected annual rents. Figure 1 summarizes the consensus (see above or NRDC 1987) among managers in the Pamlico region.

This consensus (Figure 1) implies three basic relationships:

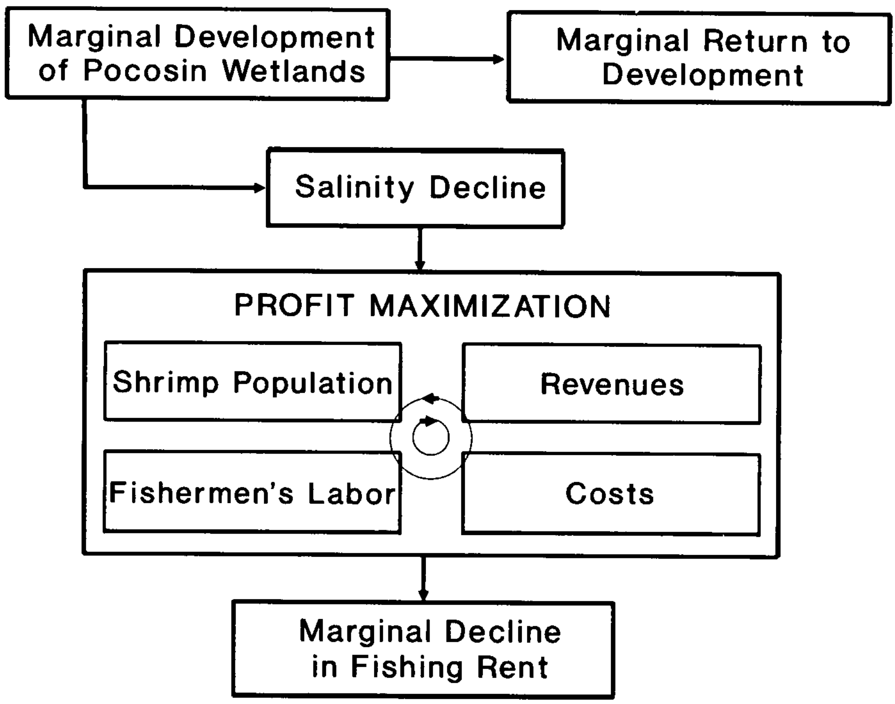

Figure 1. Outline of Empirical Model: Pocosin Wetland Development Impacts Estuarine Shrimp Fishery

$$\pi = f(XFB, L) = \text{total revenue} - \text{total costs}; \tag{5}$$

$$XFB = g(SAL); \tag{6}$$

$$SAL = h(POC); \tag{7}$$

where π is annual "profit" or quasi-rent, XFB indexes the shrimp population X, L represents craft-days of fishermen's labor, SAL is salinity in estuarine nursery areas, and POC is the acreage of pocosin wetlands on the Pamlico-Albemarle Peninsula. Rent (5) estimates renewable resource benefits, R, as a function of X and L; here π represents R and XFB proxies for X in (2). The combination of (6) and (7) empirically represent (1), where POC denotes the stock of undeveloped pocosin wetlands ($E = (E^H, E^N)$ above).

Consistent with (2), the results below derive from a simulation of Pamlico's shrimp fishery under efficiently restricted access. Benefits based on restricted access tend to estimate an upper limit to benefits under open access. The current study assumes labor may easily switch to competing fisheries, so the marginal unit of labor earns the opportunity cost available elsewhere. Available data do not identify specific vessels, so that the empirical harvest equation (below) is for a "representative vessel."[4]

The Data and Key Variables

The NC Division of Marine Fisheries and the NC Land Resources Information System (Lukin and Mauger 1983) provided the majority of data (see Swallow 1988). These weekly data included: price, catch, and effort measures for 1978–86; a fishery-independent survey of juvenile shrimp abundance and salinity levels in Pamlico's nursery areas, 1979–86; and the acreage of land in various uses, based on analysis of 1980–81 air photos and tabulated for each USGS topographic map of the Pamlico-Albemarle Peninsula. Additional data included daily rainfall records (Wiser 1983) and USDA price indices for shrimp and farmed meats (1978–84). Dollar values were adjusted to December 1986 using the producer price index (monthly series).

Pocosin wetlands may contribute differently due to their geographic location relative to the estuarine salinity regime. The Peninsula extends eastward, serving as the northern boundary of Pamlico Sound. Thus, wetlands on the south shore may buffer freshwater inflows to shrimp nursery areas, but the southeast shore's wetlands are close to inlets for undiluted seawater and may, therefore, be less critical (see Epperly and Ross 1986; Giese et al. 1985). These geographic concerns defined, for this study, wetland quality relative to shrimp, with a stock of "normal" wetlands near the southeastern shore and a "high quality" stock near the southwestern shore.

The shrimp fishery includes three Penaeid species, but brown shrimp (*Peneaus aztecus*) are the focus of this study. Brown shrimp comprise the bulk of total catch (always >65%; ≥80% in 7 of 9 years of available data) and are particularly sen-

[4] This limitation introduces a downward bias in the estimated fishery rents, since rents accruing to more skilled fishermen cannot be estimated separately (see Copes 1970).

sitive to salinity changes (see Street and McClees 1981). Brown shrimp inhabit nursery areas exclusively during the spring wet-season when estuarine salinity is sensitive to previous wetland drainage and juveniles of other species are largely absent (Heath 1975, Williams 1955). Fishermen and consumers do *not* differentiate among species, so that species-specific harvest functions are inappropriate. The empirical model estimates expected total harvest of all Penaeids.

The data permitted calculation of a population index for juvenile brown shrimp. The annual index, XFB, is simply the average of all trawl samples (no. brown shrimp caught per min.)[5] taken by NC Division of Marine Fisheries (NC DMF) in juvenile nursery areas during weeks 18 to 25 of each year. This average included samples which contained zero brown shrimp, but only considered sampling locations in known shrimp habitat (locations producing at least one shrimp over the eight years (1979–86) of available data).

Parameter Estimates and Simulation Results

A chain of marginal effects across (5)–(7) empirically links irreversible wetland development and renewable resource benefits (Fig. 1):

$$(\partial R^*/\partial X)(\partial X/\partial E) = (\partial \pi/\partial XFB)(\partial XFB/\partial SAL)(\partial SAL/\partial POC). \qquad (8)$$

This section estimates these linkages.

Rent Function

The empirical model estimates and sums weekly quasi-rents to obtain annual rents π for the brown shrimp index existing in each year. This calculation quantifies the main cell in Figure 1. Weekly rents are:

$$\pi_i = p_i\, C_i - w_s\, L_i = p_i\, k_i\, (XFB)^{\varphi}\, (L_i)^{\beta} - w_s\, L_i \qquad (9)$$

where i indexes weekly observations; p_i is the price per pound (heads-off) for shrimp of average size; C_i is the catch of all shrimp; w_s is the marginal opportunity cost of a craft-day during sub-season s; L_i corresponds to fishermen's labor in 24-hour craft-days; k_i (defined in Table 1) measures exogenous fluctuations in shrimp abundance as shrimp migrate from nurseries through Pamlico Sound to the Atlantic (see Appendix); XFB is the brown shrimp index for the current year.

Table 1 summarizes parameter estimates for the catch equation (implicit in (9)). The opportunity cost of a craft-day was estimated, following Bell (1986), by recognizing that the data were generated under open access, so that total revenues would approximate total costs. Using Bell's (1986) approach, Swallow (1988, p. 137) estimated w_s at \$714, \$1131, \$959 per 24-hour craft-day for early, middle, and late seasons (before July, July to September, after September, respectively).[6] The

[5] The sample only includes sites where NC DMF uses a 3.2 m head-rope for towing the trawl. DMF uses this head-rope in shallower, upstream nurseries where land-use changes have the most direct effect.

[6] The physical productivity of a craft-day is assumed constant, but its opportunity cost varies, e.g. due to seasonality in competing fisheries. Preliminary regressions examined

Table 1
Estimated Weekly Harvest Equation (N = 222, 1979–1986).[a]

Variable	Model 1 OLS	Model 1 GLS[c]	Model 2[b] OLS	Model 2[b] GLS[c]
Components of k_i:				
Intercept	3.22	3.28	3.20	3.26
(ln[A])	(0.190)	(0.154)	(0.188)	(0.144)
Juv. white	0.186	0.198	0.187	0.199
shrimp index	(0.0464)	(0.0329)	(0.0462)	(0.0322)
(XFW)[d]				
No. of	0.341	0.399	0.381	0.419
landings (N_i)[e]	(0.0888)	(0.0724)	(0.0595)	(0.0417)
Lag catch	0.144	0.130	0.152	0.133
(C_{i-1})	(0.0279)	(0.0222)	(0.0247)	(0.0180)
Juv. brown	0.315	0.312	0.300	0.310
shrimp index	(0.0529)	(0.0334)	(0.0460)	(0.0295)
(XFB)				
Fishermen's	0.744	0.709	0.700	0.690
labor (L_i)	(0.0850)	(0.0712)	(0.0460)	(0.0295)
F	870.	1162.	1091.	1460.
R-square	0.953	0.964	0.953	0.964
SSE/216	0.18517	0.18585	0.18462	0.18610

[a] Catch in week i is $C_i = A\,(XFW)^a\,(N_i)^b\,(C_{i-1})^c\,(XFB)^\varphi\,(L_i)^\beta$. Index i denotes weekly observations, XFW and XFB are annual observations. N and L were standardized by Swallow (1988, pp. 112–21). S.e. given in parentheses; all t-tests and F-tests were significant ≤ 0.001. The equation was estimated in log form so the estimate of ln(A) produces a biased estimate of A (Goldberger 1968) (see Appendix).

[b] Restricts $\varphi + \beta = 1$. Lagrangian multiplier test of this restriction was not significant, even at 0.50 (1 df t-statistic = 0.604 for OLS and 0.202 for GLS).

[c] OLS regressions were heteroscedastic, assuming the multiplicative model of Harvey described by Judge et al. (1985, pp. 439–441). This model assumed the logged-variance for each observation was a function of an intercept and $\ln(L_i)$. The test statistic was significant at 0.01 (1 df Chi-square = 20.83 and 24.34 for models 1 and 2, respectively).

[d] Calculated by methods used for XFB (see text), but with juvenile samples from weeks 28 to 35 for white shrimp (*P. setiferus*). A similar variable for pink shrimp (*P. duorarum*) was not significant at 0.05.

[e] Number of times craft docked and unloaded shrimp.

Appendix provides more details. The remaining discussion uses the GLS Model 2 (Table 1) to estimate π_i in (9), because that model is jointly concave in XFB and L. In Model 2, concavity is assured by the statistically insignificant restriction that $\varphi + \beta = 1$ ($P > 0.50$; Table 1).

Maximum annual rents for π in (5) are the sum of the maximized weekly rents

seasonal effects in the catch equation, but results were inconsistent with ecology and produced trivial differences in the model fit ($R^2 > 0.95$ in Table 1) (see Swallow 1988, pp. 122–127).

for π_i in (9);[7] this value was calculated for a typical year, using means for exogenous variables. Weekly rents are maximized by L_i^* such that:

$$\beta\ p_i\ k_i\ (XFB)^{\varphi}\ (L_i^*)^{\beta-1} = w_s. \tag{10}$$

Substituting L_i^* into (9) and summing over i, one obtains the annual value of marginal product of the brown shrimp population index (VMP_B):

$$VMP_B = \partial\pi^*/\partial XFB = \varphi\ \Sigma_i\ p_i\ k_i\ (XFB)^{\varphi-1}\ (L_i^*)^{\beta} = [\varphi/XFB]\ (TR), \tag{11}$$

where π^* without a subscript denotes annual maximized rents as in (5) and the last equality derives from the definition of annual total revenues (TR) (with (9)). VMP_B in (11) is a simple function of revenues, the shrimp index, and the elasticity of rents (φ) with respect to the index. For the mean observed shrimp index XFB (12.98), parameters in Model 2 (Table 1), and mean exogenous conditions (Appendix), VMP_B equals \$131,952/index point/year.[8] With $\varphi + \beta = 1$, simple analytical or numerical analyses confirm that VMP_B remains constant.

Shrimp versus Salinity

A two-step process estimated (6), the effect of salinity on the shrimp index. The statistical step estimated brown shrimp abundance in estuarine nurseries as a function of salinity. The second step used statistical results to estimate the change in the brown shrimp index XFB and fishery benefits that might result from marginal reductions in salinity. The results quantify the effect of salinity on rents (Figure 1). This section summarizes the key results for the present discussion (details are in citations below).

Using data from monitoring sites in shrimp nursery areas, the statistical step regressed the number of juvenile brown shrimp caught per minute (NBRW) against salinity, water temperature, and week (weeks 18–25; 1979–86). For samples with at least one brown shrimp, final results yielded:

$$NBRW = k_B + 1.476\ SAL \tag{12}$$

where the intercept, k_B, represents independent variables that proxy for salinity-independent effects on juvenile shrimp production.[9]

This regression (12) allows an estimate of the relationship between the shrimp index and salinity because the typical (mean) shrimp index XFB is the mean of NBRW across years:

[7] Kellogg et al. (1986) analyze the effects of discounting and shrimp growth. These effects are beyond the scope of this paper.

[8] Based on GLS Model 1 (Table 1), estimated VMP_B is only \$99,893.

[9] Since 449 of 1074 observations contained zero brown shrimp, the regression followed a censored-data approach (Lee et al. 1980). The final regression is significant ($P < 0.01$; with $R^2 = 0.12$, with a significant salinity coefficient (s.e. = 0.4862; $P < 0.01$). k_B depends on dummy variables for each year, water temperature, and week number; salinity is in parts per thousand. A quadratic term for SAL was rejected ($P > 0.05$) by an MSE test (Toro-Vizcarrondo and Wallace 1968). Swallow (1988, pp. 192–99) gives details.

$$XFB = [\Sigma_y\ FB_y \cdot (\Sigma_j\ NBRW_{jy})/N_y]/8 \tag{13}$$

where N_y is the number of NBRW samples from year y, summation j is over those samples, summation y is over the eight years of data, and FB_y is the fraction of samples in year y with at least one brown shrimp. Regression (12) estimates the effect of a marginal change in salinity on NBRW, such that

$$\partial NBRW/\partial SAL = 1.476. \tag{14}$$

Using (13) and (14), the link between an across-the-board change in salinity (at all sites) and the brown shrimp index for an average year is estimated as

$$\partial XFB/\partial SAL = [\Sigma_y\ FB_y \cdot (N_y \cdot 1.476)/N_y]/8 = 1.476\ (\Sigma\ FB_y)/8 = 0.8635, \tag{15}$$

where the last equality uses the observed mean of the annual fraction of samples containing at least one brown shrimp (mean of $FB_y = 0.5850$).[10]

Slope (15) proxies for the marginal physical product of salinity in the production of juvenile shrimp (MP_{SAL}). Equation (15) with (11) permits estimation of the value of the marginal product of the salinity level in the shrimp fishery, VMP_{SAL}:

$$VMP_{SAL} = VMP_B\ (\partial XFB/\partial SAL) = VMP_B \cdot 0.8635. \tag{16}$$

For a typical year's conditions, VMP_{SAL} equals \$113,941/salinity point/year.

Salinity versus Wetland Development

A similar, two-step process estimates relationship (7), linking salinity in nursery areas to wetland acreage stocks. The result links the cells, in Figure 1, for development and salinity.

The regression step related estuarine salinity in nursery areas along the Pamlico-Albemarle peninsula to adjacent land-use (Hyde and Dare Counties, NC), including the proportion of adjacent land in general agriculture (AGRIC), forestry-forest cover (FOREST), and pocosin wetlands (WETLAND). Lukin and Mauger (1983) define these land-uses. Results showed a statistically significant ($P < 0.05$) relationship between salinity in nursery areas and the proportion of adjacent land in three land-use categories:

$$\begin{aligned} SAL = k_{SAL} &- 13.86\ AGRIC - 10.08\ FOREST \\ &- 5.444\ WETLAND - 5.104\ SESHORE \cdot WETLAND \end{aligned} \tag{17}$$

where the intercept, $k_{SAL} > 0$, incorporates variables that describe the hydrologic

[10] See Swallow (1988, p. 205). This calculation assumes $\partial FB_y/\partial SAL = 0$, which is consistent with the interpretation that the presence of brown shrimp in a sample proves habitat suitability at a site and that a marginal salinity change will not eliminate the whole site as habitat. Alternative assumptions would not leave $\partial XFB/\partial SAL$ constant, but a numerical procedure using predicted values from the statistical model can handle alternative assumptions.

Table 2

Parameters for Estimating the Impact of Wetland Conversion on Estuarine Salinity and Value of Marginal Product (VMP) of Preserving Pocosin Wetlands Rather than Converting Acres to Agriculture or Forestry-Forest Land-Uses.[a]

	Conversion Impact on Salinity			
Conversion of Wetlands to:	Symbol	Estimate	S.e.	VMP
Agriculture				
Normal quality wetlands	$\beta^{N}{}_{AGR\text{-}WET}$	3.312*	0.936	0.282
High quality wetlands	$\beta^{H}{}_{AGR\text{-}WET}$	8.416*	1.01	3.37
Forestry-Forest				
Normal quality wetlands	$\beta^{N}{}_{FOR\text{-}WET}$	−0.468‡	0.907	—
High quality wetlands	$\beta^{H}{}_{FOR\text{-}WET}$	4.636*	0.476	1.85
Constant of proportionality ($K^{i}{}_{WET}$) to calculate $\partial XFB/\partial WETLAND^{i}$:				
Normal quality wetlands (i = N)	0.4376×10^{-6}			
High quality wetlands (i = H)	2.054×10^{-6}			

[a] Based on equation (17) and estimated covariances.
* Significantly different from zero at $P < 0.01$.
‡ Not significantly different from zero, even at $P < 0.25$.

conditions for the year, and dummy variable SESHORE[11] equals 1 for sites near the peninsula's southeastern shore.[12]

The coefficients on the land-use variables in (17) show that converting pocosin wetlands (lowering WETLAND) to agriculture or forestry (raising AGRIC and/or FOREST) causes a *net decline* in the salinity of nearby shrimp nursery areas. Table 2 gives estimated parameters for the net effects on salinity. For example, converting wetlands on the southwestern shore (SESHORE = 0) to agriculture decreases mean salinity in adjacent estuarine nurseries by 8.4 units (13.86 − 5.44; see (17) and Table 2). These results are consistent with the expectation that south*western* wetlands are of "high quality" (*i.e.*, E^H) while south*eastern* wetlands are of "normal quality" (i.e., E^N).[13]

One can show that developing peninsular wetlands has an estimated impact on

[11] Southeastern shore is from longitude 75° 53′ 25″ W to 76° 07′ 30″ W.

[12] See Swallow (1988, pp. 179–87, 213–19) for variable definitions and discussion of preliminary regressions. The standard errors for (17) are, respectively, 1.912, 1.057, 1.159, and 0.7768. These estimates pertain to an equivalent GLS model which corrects for heteroscedasticity across years of data. All variables were significant ($P < 0.01$) as is the model ($F_{13,301} = 439.014$; $P < 0.001$; $R^2 = 0.9499$; $N = 315$). The intercept term is given by

$$K_{SAL} = \underset{(0.7918)}{24.81} - \underset{(0.3650)}{14.47}\ D79 - \underset{(0.2481)}{13.79}\ D80 - \underset{(0.7815)}{3.363}\ D81 - \underset{(0.2713)}{5.118}\ D82$$
$$- \underset{(0.2984)}{12.40}\ D83 - \underset{(0.2804)}{11.79}\ D84 - \underset{(0.3899)}{1.504}\ D85 - \underset{(0.08329)}{0.2170}\ RAIN\,10\ \ \underset{(0.4365)}{4.702}\ SESHORE$$

where standard errors are given in parentheses, dummy variables for each year are Dnn (nn = 1979 to 1985), and RAIN10 is the rainfall in the ten days preceding the salinity sample.

[13] Land-use variables are proportions (acreage of land-type divided by total acreage sampled, TOTACRE). E.g., converting 1 acre of wetlands to agriculture decreases the numerator of WETLAND by 1 and increases the numerator of AGRIC by 1.

the shrimp index that is proportional to the coefficients in Table 2. Doing so uses (13) and splits the summation over sampling sites j into sites near one shore of the peninsula (j:SESHORE $\rightarrow$ i; i = H,N)[14] and sites away from that shore,

$$\partial XFB/\partial WETLAND^{i} = (1/8)\Sigma_{y}\{(FB_{y}/N_{y}) \cdot [\Sigma_{j:SESHORE \rightarrow i}(\partial NBRW_{jy}/\partial SAL)(\partial SAL/\partial WETLAND^{i})_{j} + 0]\} \tag{18}$$

where the zero relates to sampling sites away from that shore; and $\partial SAL/\partial WETLAND^{i}$ (i = H,N) is estimated from (17) (*i.e.*, the βs in Table 2 with SESHORE = 1 for i = N). For agricultural development, (18) simply becomes

$$\partial XFB/\partial WETLAND^{i} = K^{i}_{WET}\ \beta_{SAL}\ \beta^{i}_{AGR\text{-}WET},\ i = H,N \tag{19}$$

where β_{SAL} is the coefficient on salinity in (12), $\beta^{i}_{AGR\text{-}WET}$ comes from (17) with Table 2, and K^{i}_{WET} captures the remaining terms in (18)[15] (see Table 2).

Valuation of Pocosin Wetlands for Shrimp Fishery

By (11) and (19), the marginal value of peninsular wetlands for brown shrimp production is estimated by:

$$\begin{aligned} VMP_{i} &= (\partial\pi^{*}/\partial XFB)\ (\partial XFB/\partial WETLAND^{i}) \\ &= VMP_{B} \cdot (\partial XFB/\partial WETLAND^{i}),\ i = H,N; \end{aligned} \tag{20}$$

where this annual value of marginal product is given in Table 2 for each wetland quality and each potential land-use that wetlands displace (*cf.* Fig. 1). For example, the highest losses to the shrimp fishery are estimated as \$3.37/acre/year for developing agriculture on wetlands near the southwestern shore (Table 2). Results also show that conversion to forestry-forest cover may only be a concern for high quality wetlands, from the perspective of protecting shrimp nurseries. Results for normal quality (southeastern) pocosins remain consistent with forestry research suggesting that modern management may maintain the hydrologic role of these wetlands (Campbell and Hughes 1991), but results for the high quality (southwestern) wetlands suggest that forestry in some locations alters wetland functions and values.

The estimates in Table 2 represent values for an average year based on the statistical parameters. Table 3 provides a sensitivity analysis in the form of upper and lower bound estimates derived by using the 95% confidence bounds for each of the three key parameters. Value estimates are most sensitive to potential estimation error in the linkage between the shrimp index and salinity, equation (12). These bounds also show that conversion of normal wetlands to forestry-forest land-uses may cause fishery losses up to \$0.22/acre-year, despite statistical insignificance of the impact on salinity.

[14] That is, j identifies sites with one value of SESHORE and that value determines quality index i.

[15] For any sampling site and for the agriculture example, $(\partial SAL/\partial WETLAND^{i})_{j} = \beta^{i}_{AGR\text{-}WET}/TOTACRE_{j}$; $TOTACRE_{j}$ is the total acreage of land in the land-use sample near site j. K^{i}_{WET} includes $\Sigma_{j:SESHORE \rightarrow i}(1/TOTACRE_{j})$.

Table 3

Sensitivity Analysis of Parameters on the Value of Marginal Product of Normal (VMP_N) and high (VMP_H) Quality Wetlands for Protection of Salinity in Brown Shrimp Nursery Areas[a]

Conversion of wetlands:	Using S.E. of (equation no.)[b]	VMP_N Upper	VMP_N Lower	VMP_H Upper	VMP_H Lower
To agriculture	φ (9)	0.335	0.230	4.00	2.74
	β_{SAL} (12)	0.464	0.100	5.54	1.19
	$\beta_{AGR\text{-}WET}$ (17)	0.439	0.126	4.16	2.58
	All three	0.856	0.036	8.12	0.743
To forestry-forest[c]	φ (9)	—	—	2.20	1.51
	β_{SAL} (12)	—	—	3.05	0.657
	$\beta_{FOR\text{-}WET}$ (17)	0.112	−0.191	2.23	1.48
	All three	0.218	—	4.35	0.427

[a] Upper and lower bounds calculated using, respectively, +1.96 or −1.96 times the standard error of estimated parameters, as indicated.

[b] β_{SAL} is the coefficient on SAL in (12); $\beta_{AGR\text{-}WET}$ and $\beta_{FOR\text{-}WET}$ are defined in Table 2 using coefficients in (17). "All three" bounds are calculated using the standard errors of φ, β_{SAL}, and either $\beta_{AGR\text{-}WET}$ or $\beta_{FOR\text{-}WET}$.

[c] Some estimates are omitted since results in (17) cause the base value of $\beta_{FOR\text{-}WET} < 0$ and $\beta_{FOR\text{-}WET}$ is statistically insignificant, so some bounds provide no meaningful information.

Finally, these results refute the common notion that wetlands may be valued as a homogeneous group. The results show variation in value, despite the absence, in the analysis, of a detailed accounting of the various ecological types of pocosin wetlands (Lukin and Mauger 1983; Richardson and Gibbons 1993). The variation here depends not only upon geographic location, which may correlate with ecological types, but also upon the type of developed land-use contemplated. The results illustrate that wetlands values depend not only on their role in the ecosystem but also on the role that the alternative land-use would play in the ecosystem.

Tradeoff Assessment and Implications

This section assesses the preservation and development tradeoffs in pocosin wetland development by combining the empirical results with Heimlich and Langner's (1986) analysis of returns to agriculture. The objective is illustrative, rather than prescriptive, highlighting advantages of a resource-theoretic framework and a stepwise empirical approach.

Some assumptions are necessary to adjust the available results for a resource-theoretic framework. First, on the preservation side, we assume the results for shrimp are indicative for other fisheries and that economic impacts on other Pamlico fisheries are proportional to their gross dock-side value;[16] then impacts of salinity or wetland changes on shrimp represent 25% of total impacts of agricul-

[16] Some precedent for this type of assumption exists in Kahn and Kemp (1985); see also arguments in Gupta and Foster (1975).

tural development. Second, on the development side, we build on Heimlich and Langner (1986) and assume returns to wetland conversion decline as development proceeds eastward, moving further from mainland transportation networks. Finally, an 8% discount rate is assumed.

Empirical results show that, in Pamlico's estuarine nursery areas, the marginal loss due to an across-the-board reduction in average salinity is about $114,000 annually or $1.4 million in present value for shrimp, or about $5.7 million in present value for all fisheries. Since policy makers (NRCD 1987) link coastal zone development to salinity reductions, these estimates justify debate over *net* benefits of development.

This debate has emphasized wetland development for agriculture. Such development annually impacts shrimp fisheries by $3.37 1986-dollars per acre of high quality wetlands developed and $0.28 per acre of normal quality wetlands, with respective present values of $42.13 and $3.50 per acre. These estimates initially may appear low, but they apply to diffuse and indirect impacts. The annual marginal loss is comparable to previous studies: losses from direct destruction of habitat for Florida blue crabs are about 0.30/acre-yr 1975-dollars (Lynne *et al.* 1981); wetlands that mitigate hurricane damage provide benefits of 0.40/acre-yr 1980-dollars (*cf.* Farber 1987). In 1986-dollars, these comparison values are $0.44–$0.48/acre-yr. Finally, Heimlich and Langner (1986) identify pocosin development as a marginal investment, so that our estimated fishery losses may be large relative to development values.

The stepwise empirical model offers one significant lesson. Pocosin wetlands comprise one resource that benefits the shrimp fishery. Yet the intermediate results indicate a real potential for *general* coastal zone development to cause substantial *aggregate* losses via salinity. A localized focus on specific types of development (*e.g.*, pocosin development) may miss significant impacts from development in the full watershed of an estuary.

Following the resource-theoretic framework, we now assess the net opportunity cost (NOC in (4)) of agricultural development. Adjusting the per-acre wetland values (Table 2) to acknowledge all Pamlico fisheries, the preservation value becomes $13.48 and $1.12 per acre annually for high and normal quality pocosins. Heimlich and Langner (1986) suggest that converted wetlands sell for $1350 per acre, while Barnes (1981) estimates acquisition and development costs around $1190. However, Heimlich and Langner (1986) estimate "typical" development costs above $1500, suggesting that more than half of remaining wetlands are not economic for agriculture. Our illustration assumes 20% of wetlands remain economic for development, with $C^i(E^i) > 0$ in (4) ($i = H,N$); this acreage includes 2809 acres of high quality, southwestern wetlands and 11,009 acres of normal quality, southeastern wetlands (13,818 acres total). The illustration also assumes the first acre earns an annualized return of $12.8 (*i.e.*, $rC^H(2809) = 0.08 \cdot [1350–1190]$) and the return declines at a constant rate per acre so that $rC^N(E^N) = 0$ for acre number 13,818. These assumptions yield example equations for the returns to development:[17]

$$C^H(E^H) = 160 - 0.01158 \cdot (2809 - E^H) \tag{21a}$$

[17] Available data on returns to wetland conversion omit geographic location. Readers may evaluate other assumptions at their discretion.

$$C^N(E^N) = 127.47 - 0.01158 \cdot (11009 - E^N), \tag{21b}$$

where it is assumed that the return to the last marginal unit of high quality wetlands equals the return to the first marginal unit of normal quality wetlands; that return is \$10.20 annualized (0.08 times \$127 present value).

In this example, the "efficient" policy would be to preserve all high quality (southwestern) wetlands because, despite their higher value for agriculture, their value for fisheries is higher still and the net opportunity cost of development is negative ($N^H < 0$). In contrast, the policy would allow development of normal quality (southeastern) wetlands, but would halt development when the marginal net value of development fell to an annualized \$1.12 per acre (or \$14 in present value), where the net opportunity cost of development just equals zero ($N^N = 0$). In this example, developers use 9800 of 11,009 acres of southeastern wetlands, leaving 1209 acres preserved.

The example compromises development and preservation interests. Developers forego their most profitable wetlands because these same wetlands are most valuable to fisheries. However, preservationists lose 89% of the normal quality wetlands that development threatens,[18] preserving 11%.[19]

Concluding Summary

A resource-theoretic approach to development and preservation tradeoffs merges renewable and nonrenewable resource theories, highlighting that both preservation and development contribute positively to social welfare. The framework is applied to preservation and development tradeoffs between agricultural development of pocosin wetlands and its impact on estuarine fisheries. An illustrative assessment supports preservation of wetlands that are most attractive to both development and fishery sectors, while development of some less highly valued wetlands may be efficient.

Furthermore, a stepwise approach to link freshwater wetland development and estuarine shrimp production reveals a potential for substantial welfare losses if estuarine salinity declines across-the-board. This result encourages research to identify impacts of development throughout the estuarine watershed. This stepwise approach also offers a number of stages where future biophysical-economic models may enter the evaluation. Such flexibility may be important as methods for estimation of total values improve to account for non-consumptive uses (*e.g.* Costanza *et al.* 1989; Whitehead 1993). Finally, supporting Crocker and Tschirhart (1992), empirical results reveal that tracing human impacts through ecosystem linkages affects resource valuation: wetland values depend not only on their ecological type, but also on factors such as their geographic location and the land-use (agriculture versus forestry) in their developed state.

[18] Recall the example assumes 80% of southeastern wetlands are not threatened and are preserved by default.

[19] Given the linearity in (21) and the \$160 present development value of the first acre, both the preservation of all high quality wetlands and the proportion of normal wetlands preserved are *invariant* to alternatives (e.g. 50%) to the assumption that 20% of all remaining wetlands offer positive marginal returns to development.

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Appendix

This appendix provides details on the data and the parameters used to simulate potential rents from the Pamlico Sound, NC shrimp fishery (Table A.1). This material will facilitate replication of the empirical results.

The empirical model treats the number of times craft landed shrimp (docked and unloaded), N_t, as one index for shrimp migratory timing based on a bioeconomic argument detailed in Swallow (1988, pp. 121–40). In the context of Babcock and Mundy (1985), bioeconomic support exists for using observed catch (C_{t-1}) as a simple proxy for migratory timing since the data derive from time-invariant harvest regulations (open access).

Finally, based on Williams' (1955) life history research, the empirical analysis uses a large white shrimp index (XFW) as an indicator of prolonged high salinity conditions in nursery areas; the observed mean XFW was 0.629. If salinity is high in nurseries during summer and early fall, the survival rate of white shrimp population will be higher. This event indicates a better year for brown shrimp production since spring salinity levels are correlated with late-season salinity levels (Heath 1975).

Catch effort data covered 1978–86, with annual series beginning in weeks 17–25 and ending in weeks 48–52 (see Swallow 1988, pp. 105–6). Shrimp indices only covered 1979–86. Estimates in Table 1 derive from 1979–86 data, while means in Table A.1 include available 1978 data. Since 1978 produced a below average shrimp harvest, including 1978 in the estimate of mean conditions lowers the estimate of rents and marginal values.

The expected price in week i, p_i (Table A.1), was estimated as a function of a national price index (available from U.S. National Marine Fisheries Service for 1978–84) and shrimp size. In turn, average size was estimated as a function of week number and seasonal dummy variables. See Swallow (1988, pp. 140–48). For

Table A.1

Weekly Mean Values Used to Simulate Exogenous Migratory Timing of Shrimp (N_t, C_{t-1}) and Ex-Vessel Price (p_t) (December 1986-dollars, heads-off) in Quasi-rent Equation (9) with Table 1

Week (t)	No. of landings (N_t)	Catch (lbs) (C_t)	Price ($/lbs) ($p_t$)	Week (t)	No. of landings (N_t)	Catch (lbs) (C_t)	Price ($/lbs) ($p_t$)
17	0.4	42.7	3.145	35	160.6	84325.4	3.870
18	0.9	72.7	3.217	36	121.8	66180.7	3.932
19	1.3	54.6	3.078	37	103.4	52424.3	3.941
20	1.9	165.0	3.168	38	99.4	46276.8	3.940
21	3.3	320.1	3.301	39	82.9	38719.1	3.929
22	5.9	1218.6	3.271	40	82.4	40984.8	3.918
23	10.9	2459.9	3.057	41	88.3	44101.0	3.449
24	14.0	3292.4	3.141	42	72.7	33787.6	3.414
25	28.6	9860.2	3.226	43	63.0	33343.3	3.371
26	73.0	33773.3	3.309	44	63.2	28316.6	3.275
27	138.3	95070.4	3.409	45	61.8	23384.8	3.213
28	190.6	118663.1	3.490	46	42.7	12278.0	3.150
29	229.6	168633.6	3.565	47	25.8	8410.0	3.083
30	223.0	160208.0	3.635	48	17.7	6121.2	3.001
31	198.5	139427.2	3.718	49	10.2	4174.7	2.930
32	230.1	162642.7	3.773	50	6.4	2266.4	2.854
33	205.0	120820.1	3.820	51	2.3	742.9	2.778
34	164.9	93128.4	3.857	52	1.1	72.1	2.703

simplicity, this study used the weekly average size of shrimp rather than disaggregated size classes; however, statistical results were consistent with those of Kellogg et al. (1986). Furthermore, USDA indices for prices received by farmers for hogs and for all meat animals made no statistical improvements.

For the simulation results, the intercept in Table 1 was adjusted for bias by modifying Goldberger's (1968) procedure. This modification approximates the regression variance used by Goldberger (1968) with the sum of squares of the regression errors from the GLS model, divided by error df. The correction multiplies A (Table 1) by 1.0974 and 1.0975 for models 1 and 2, respectively. While the modification is *ad hoc* in the presence of heteroscedasticity, omitting the correction biased rents downward by >200%. (See Swallow 1988, pp. 154–56).