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# VALUATION AND MANAGEMENT OF WETLAND ECOSYSTEMS

ROBERT COSTANZA [1], STEPHEN C. FARBER [2] and JUDITH MAXWELL [3]

[1] *Coastal and Environmental Policy Program, Center for Environmental and Estuarine Studies, Chesapeake Biological Laboratory, University of Maryland, Solomons, MD 20688-0038 (U.S.A.)*
[2] *Economics Department, Louisiana State University, Baton Rouge, LA 70803 (U.S.A.)*
[3] *School of Natural Resources, 2021 Coffey Road, The Ohio State University, Columbus, OH 43210-1085 (U.S.A.)*

## ABSTRACT

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We recently completed a study of wetland values in coastal Louisiana that employed both willingness-to-pay and energy analysis-based methodologies and were able to bracket a range of values within which we feel fairly confident the true value lies. However, a large amount of uncertainty remains. Our current estimates of the total present value of an *average* acre of natural wetlands in Louisiana are US \$2429–6400 per acre (assuming an 8% discount rate) to \$8977–17000 per acre (assuming a 3% discount rate). At the lowest value, the current annual rate of loss of Louisiana wetlands (50 sq miles per year) is worth about \$77 million. At the largest value it is worth about \$544 million.

In this paper we (a) discuss the fundamental theoretical and practical problems underlying natural resource valuation; (b) summarize our methods and findings for Louisiana wetlands; and (c) elaborate on some of the more recalcitrant problems attending applied natural resource valuation, including discounting and dealing with uncertainty and imprecision.

The discount rate makes more difference in the final result than any other one factor, and yet there is much disagreement about the appropriate approach to discounting natural resources. We discuss the discounting problem as applied to natural resources and argue for lower discount rates for valuing renewable natural resources than apply for other aspects of the economy.

It now seems clear that no reasonable amount of effort will produce very precise estimates of wetland values, and we suspect this is also the case for several other classes of natural resources. We elaborate a Wetlands Assurance Bonding system to address these problems.

## BACKGROUND

### *Imperfect markets and prices*

In market economies, prices are relied on to provide signals regarding resource scarcity. These prices are considered to provide a true measure of

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economic value only if the market is characterized by a large number of buyers and sellers and by private property rights which are enforceable and transferable. The consumption choice must be made from a basket of 'rival' goods and services, that is, the choice to consume one good comes at the expense of some other good. If these conditions hold, prices can be relied on to direct the allocation of scarce resources to their highest valued use. Unfortunately, for the many ecosystem goods and services that humans *do* care about, such markets are of limited usefulness and/or fail to perform efficiently. The absence of efficient markets for these outputs is the major cause of what can be called 'inefficient' habitat modification. This is the conversion from those land use activities that have large, real, but uncapturable benefits (i.e., natural ecosystems) to land use activities that have smaller but capturable income streams. In this most prevalent case, individual preferences diverge from those of society as a whole and collective action is warranted.

Even when the preferences for the resources provided by existing natural ecosystems are strong, their common property nature makes it impossible to exclude those who do not pay for enjoying or using them. As a result there is no incentive to conserve such resources and overuse and even exhaustion can occur when utilization or harvest rates exceed the population growth rates of species, as in a fishery, or rates of forage production in common grazing areas. For some resources, such as camping sites and scenic views, utilization levels result in congestion and subsequent declines in resource quality (Cicchetti and Smith, 1973). For congestible goods, exclusion results in efficient use only if consumers pay for their use in accordance with their own valuation for that use plus congestion costs imposed on others. Because of obstacles to both collecting payment and in eliciting true evaluations, public or collective management is usually required.

*Lack of provision for equity and sustainability*

Another deficiency of markets is that the choices and prices observed are a function of types and levels of personal endowments. The distribution of endowments and the relative importance imparted to these endowments in the marketplace can cause significant bias in environmental valuations from the individual, societal and regional viewpoint. For example, threatened species will be valued very differently by the local population in poor countries characterized by dependence on natural resource trade to secure foreign exchange and in rich countries characterized by use of imported natural resources. Unfortunately, when consumers and producers are geographically separated, consumers are able to internalize the benefits of natural resource trade while externalizing the cost of habitat destruction.

Markets may also fail to encourage ecologically sound management when very long time horizons are involved, resulting in intergenerational inequity and irreversible damage to ecosystems. The choice of interest rate is critical in determining the optimal harvest rate for renewable resources and in turn determines their renewability. Clark (1973) has shown that the high discount (or interest) rates used by individuals to assure short-term profit maximization may cause overexploitation and exhaustion of species characterized by slow growth rates. However, even if the discount rate were zero, i.e. future values equal current values, the short lifespan of humans will result in management practices which favor current values and incomes, even if such practices have a potential to cause high social costs to future generations. For example, fluorocarbons permit packaging under pressure and a finer spray than that provided by the alternative pump spray. These benefits are directly reflected in market prices and are quite modest when compared to the catastrophic cost which would result from ozone depletion. Page (1978) contends that the inability to estimate the probability of ecocatastrophy creates indifference to the huge asymmetry between current benefits and the potential for future catastrophic costs. This makes even political solutions difficult.

Where the allocation of scarce ecological resources is concerned there will invariably be a clash between individual preferences and social benefits. Markets are just one of many arenas for resolution of these disputes. Public resource management, legislation, and regulatory agencies offer alternative methods or forums for directing management decisions. In spite of the absence of market prices, environmental decision making increasingly emphasizes the quantification of benefits and costs. This has resulted in a proliferation of new resource value concepts and valuation methods.

*Willingness to pay and extending existing markets*

For the individual, one estimate of the economic value of an increment in any good or service is the maximum amount that he or she is willing to pay (WTP) for it. Conversely, the value of a decrement is the minimum amount that the individual is willing to accept (WTA) for it. The prices formed in well functioning markets are one source of WTP and WTA estimates of marginal increments or decrements of goods and services. Where markets fail to provide appropriate measures of environmental values, the WTP and WTA concepts of economic value are not invalidated, but alternative 'pseudomarkets' must be used to elicit these values from individuals.

The notion that an alternative chosen will be at the expense of some opportunity foregone is central to economic decision making. For example, the cost of providing a scenic view can be directly derived from the net value of outputs and uses foregone such as timber harvest and dispersed grazing. This is referred to as an *opportunity cost*. For scenic view preservation to be economically efficient, the scenic view must be preferred over other uses. In other words, its value must exceed the cost of providing it, including opportunity costs.

Ecological goods and amenities are valued by individuals for a variety of reasons: *utilitarian (or use) value* refers to the value of using an ecosystem's products and amenities to derive both current and future benefits; these benefits include commercial outputs such as timber, outdoor activities and experiences, wildlife, and aesthetics (for examples of raw material evaluation see Bartlett (1984), who discusses valuation assumptions and methods for range forage). Individuals may also be willing to pay now for the option of using a resource in the future. Such an option price includes an amount equivalent to the expected use value plus a premium, similar to a risk premium, which a person would pay over and above the expected use value. This premium is referred to as *option value*, and is due either to uncertainty surrounding the individual's preferences or to uncertainty regarding the price or availability of the resource. This premium may be positive, negative, or zero (as in the case of preference uncertainty) but it will always be positive in the case of supply availability for a risk averse person – see Greenley et al. (1981), Bishop (1982) and Brookshire et al. (1983) for the theory and empirical studies of option value. The passage of time will likely reduce the uncertainty surrounding resource usefulness. When resource use is irreversible, individuals would be willing to sacrifice current irreversible use until uncertainty about its cost has been reduced. They would be willing to pay for increased information. This payment is termed *quasi-option value* (Arrow and Fisher, 1974; Conrad, 1980). It is not attributable to risk aversion, like option value, but is due to the value of information; this value arises in the case of resource use decisions that create irreversible damages, such as species extinction or large-scale deforestation. A final, pure non-use value is what a person may be willing to pay simply to know that a resource exists even when there is no intention of use. This *existence value* has nothing to do with preserving options for future use or paying to delay use until more information is available (see Randall and Stoll, 1980, and Brookshire et al., 1983).

In practice, the measurement of these value concepts has remained difficult and largely limited to the valuation of environmental commodities and amenities which produce direct benefits to humans. An alternative approach is Norton's (1986) concept of *contributory value*, which assigns value to environmental resources not due to their *direct* value to humans, but according to their *indirect* role in maintaining and accentuating the ecosystem processes which support these direct benefits. These include the maintenance of atmospheric and aquatic quality, the amelioration and control of climate, flood control, the maintenance of a genetic library, and the supportive role of food webs and nutrient cycling. Contributory value recognizes both the long time horizons involved in many ecosystem processes and the synergism which can result from the interaction of two or more species creating benefits of which neither is individually capable.

Though empirically elusive, contributory value does provide a useful framework for conceptualizing how natural ecosystems might be evaluated. However, as Randall (1986) contends, human preferences are focussed more on life forms than on life processes. This bias is further distorted by the fact that humans, in general, will assign higher preferences to species with commercial value, to wild relatives of domesticated species and to those which are most familiar and/or easy to empathize with, such as large mammals. Lovejoy (1986) refers to this bias against invertebrates as vertebrate chauvinism, while others point to interspecies inequity (Costanza and Daly, 1987). If it is accepted that each species, no matter how uninteresting or lacking in direct usefulness, has a role in natural ecosystems (which do provide many direct benefits to humans), it is possible to shift the focus from the imperfect perceptions of individuals to the contributory value of ecosystems as expressed through their ecological relationships. One might argue that this contributory value is an estimate of the value individuals would place on environmental services if they were fully informed about the functioning of the environment in their behalf.

*Ecosystem function and economic value*

Assessing the contributory value of ecosystems involves the ability to understand and model the ecosystem's role in an integrated ecologic–economic system and its response to perturbations. The models must be at a level of detail and resolution that allows the assessment of impacts (marginal products) on economically important ecosystem commodities and amenities. Several types of ecological modeling can be used for this purpose, which we define under the general heading of 'ecological–economic' models. They range from relatively simple, static, linear input–output models (Isard, 1972; Hannon, 1973, 1979; Costanza and Neill, 1984; Costanza and Hannon, 1990) to multiple regression models (Farber and Costanza, 1987) to more sophisticated nonlinear, dynamic spatial simulation models (Costanza et al., 1986). Braat and Van Lierop (1986) provide a summary of ecological–economic models currently in use.

The point that must be stressed is that the economic value of ecosystems is connected to their physical, chemical, and biological role in the overall system, *whether the public fully recognizes that role or not*. Standard economics has too often operated on the assumption that the only appropriate measures of value are the current public's subjective preferences. This yields appropriate values only if the current public is *fully informed* (among a host of other provisios). The public is most likely far from being fully informed about the ecosystem's true contribution to their own well being, and they may therefore be unable to directly value the ecosystem's services (Constanza, 1984). However, scientists may be able to derive estimates of the values that a fully informed public would produce by analyzing the structure and function of ecosystems. Economists also tend to assume that local optimizing of independent individuals will produce optimal results for the society. We have noted that this only works when there are perfect markets and that perfect markets are the exception rather than the rule in the natural resource area. Therefore, once appropriate values for ecosystem services are derived, that information must be inserted into the decision making process in order to correct the market signals.

## WETLAND VALUATION

This section provides an example of the methods and problems in deriving empirical estimates of natural wetland values (details are available in Costanza and Farber, 1985a,b; and Farber and Costanza, 1987). Two different wetlands valuation techniques were used: willingness-to-pay (WTP) valuation and energy analysis (EA) valuation. These methodologies are thoroughly explained elsewhere (Costanza and Farber, 1985a,b; Farber and Costanza, 1987). We provide a brief summary here.

WTP valuation refers to valuing the particular dimensions of benefits or projects by determining society's willingness-to-pay (WTP) for those particular benefits. It requires a listing of the types of benefits and an estimate of the WTP for each one. Our analysis of the WTP for wetlands concentrated on four major categories of benefits of wetlands: commercial fishing and trapping, recreation, and storm protection. Waste treatment benefits are partially included in the other benefit estimates to the extent that water quality affects recreation, fishing, and trapping values. We were not able, as part of this analysis, to place values on the existence and option value of wetlands.

The methodology for estimating commercial productivity consisted of estimating the marginal productivity of an acre of wetlands. Our estimates concentrated on the following commercial products: shrimp, menhaden, oysters, blue crab, and furs. The critical problem in estimating the marginal productivity of wetlands was to separate the effect of human effort from the effect of the intrinsic wetland productivity, when we can only observe the total effect as reflected in harvests. Failure to make this separation results in a potentially very large overestimate of the contribution of the wetlands to commercial production. A second problem in this estimation procedure is to distinguish between the average and marginal productivity of the wetlands. This is important if there exist decreasing returns to wetlands productivity (i.e. if the productivity of a unit area of wetlands depends on the amount of remaining wetlands). Perhaps average productivity is appropriate for valuing very large wetlands projects, but valuation of small projects should use marginal productivity.

The estimation of WTP for recreational value is also complex. Two techniques can be used to make this valuation. First, one can simply ask recreational users what they would be willing to pay to use the wetlands in the project area. The problem with this technique is that respondents may engage in strategic responses. For example, if they think they may have to actually pay what they say they are willing to pay, they may state a value lower than their true value. On the other hand, if they think their response may positively impact the probability of implementing a project, they may state a value higher than their true value. A second technique is to estimate recreational users' WTP based on observations of what it actually costs them to use the project area. This technique is called the travel cost method and was our primary means of assessing the recreational value of wetlands.

The value of the wetlands for hurricane protection was obtained from a methodology which determined the reduction in expected property damages in populated areas along a gradient relative to distance from the coast. In principle, people would be willing to pay for a wetlands project according to the reduction in expected property damages attributable to the project.

The energy analysis valuation technique looks at the total biological productivity of wetland vs. adjacent open water ecosystems as a measure of their total contributory value. Primary plant production is the basis for the food chain which supports the production of economically valuable products such as fish and wildlife. It is converted to an equivalent economic value based on the cost to society to replace this energy source with fossil fuel as measured by the overall energy efficiency of economic production. This technique is comprehensive and does not require a detailed listing of all the specific benefits of wetlands, but it may overestimate their value if some of the wetland products and services are not useful (directly or indirectly) to society.

We acknowledge that the overall level of precision of our current estimates of total wetland values is not as high as we would like (due mainly to cost and data constraints), especially for important categories such as option and existence values. We suggest methods for dealing with this uncertainty in a later section.

*Commercial fishery production value of wetlands.* The commercial value of wetlands for fish and wildlife harvesting is attributable to the fact that wetlands provide food and habitat necessary for the production and survival of various species. This section summarizes estimates of the value of wetlands for the harvesting of shrimp, blue crab, oysters, menhaden, and furs.

Attempts to estimate the productivity of wetlands must address the problem of separating the productivity of the wetlands themselves from the productivity of human inputs which are used in harvesting those wetlands. The problem can be outlined as follows. Suppose the harvest from a given ecological system, $Q$, is a function of the level of some environmental variable, $W$ (for example, wetland acreage), and some level of human effort, $M$:

$$Q = \mathrm{f}(M, W) \tag{1}$$

Also, the level of human effort will be a function of the price per unit of biomass harvested, $P$, the cost per unit of human effort, $C$, and the resource abundance, which, in turn, is dependent on the level of the environmental variable:

$$M = \mathrm{g}(P, C, W) \tag{2}$$

Any change in $W$ will result in a change in both $M$ and $Q$. The productivity of the environment alone is the change in $Q$, holding $M$ constant. This is referred to as the Marginal Productivity (MP) of the environmental variable. Likewise, the MP of human effort is the change in $Q$, holding $W$ constant.

Simply estimating (1) will not control for the effect of changes in $W$ on the level of human effort across observations. In addition, simply observing $Q/W$, the Average Product (AP) of the environmental variable will more than likely overestimate the marginal productivity of the environmental variable since there are probably decreasing returns to the environmental variable when human effort is held constant; i.e., AP > MP. More importantly, the average product implicitly includes the productivity of the human effort embodied in the dependent variable $Q$.

A conceptual problem also arises in the estimation of the system composed of (1) and (2). It is difficult to specify the proper environmental level ($W$) for migratory species because it is not clear which environment or set of environments they are dependent on and for how long. It is easiest to estimate this system for more stationary species because: (a) the environmental linkage is clearer; and (b) there will be more observations over which to estimate the system of equations.

The economic value of a unit of the environment, $V$, for example, an acre of wetland, is the product of MP and the dockside price, $P$:

$$V = \text{MP} * P \tag{3}$$

Estimates of this value ($V$) based on the average value of harvest per unit of environment (AP) are inaccurate since they do not account for the difference between AP and MP. Human inputs have costs and productivity, so AP will most likely overestimate the social value of a unit of the environment. Another measure has been $[(Q * B) - (M * C)]/W$, the net rent to harvesters. In a perfectly competitive harvesting industry, which is probably close to what prevails, this value is approximately zero; therefore, it will most likely underestimate the social value of a unit of the environment.

Lynne et al. (1981) developed a model of catch in which catch in year $t$, $Q_t$, depends on marsh acreage in year $t-1$, $W_{t-1}$, catch in year $t-1$, $Q_{t-1}$, and effort in year $t$, $M_t$:

$$Q_t = \beta_0 + \beta_1(\ln W_{t-1})M_t + \beta_2(\ln W_{t-1})M_t^2 + \beta_3 Q_{t-1} + \epsilon_t \tag{4}$$

The marginal productivity of marshland is then the derivative of (4) with respect to $W$. The economic value ($V$) of that marginal acre of marsh is then the marginal productivity times the dockside market value of catch.

In what follows we present summaries of our estimates of marginal productivities of marshlands with respect to commercial fishery production. We made estimates only for shrimp, menhaden, oysters, and blue crab. These species account for approximately 95% of commercial fishery harvest in Louisiana (USDC, 1983).

In our study, brown and white shrimp catch in offshore and inshore waters in Louisiana was measured using National Marine Fisheries Service (NMFS) data for subareas of Louisiana from 1963 to 1976. These data are by area of catch, not landing. There were 24 inshore subareas and four offshore areas designated by NMFS prior to 1976. Catch is measured in heads-off pounds. Effort is measured in 24-hour days. Marsh area was measured for years 1955 and 1978 by the U.S. Fish and Wildlife Service for 7.5′ quad sheets (see Costanza et al., 1983). Although data were collected on both saline and fresh marsh, this study uses the sum of the two. Area data for years between these two years were estimated by simple linear interpolation.

* acre ≈ 0.4047 ha.
* lb, pound (avdp) ≈ 0.454 kg.

TABLE 1

Estimated marginal productivities per acre of marsh for brown and white, inshore and offshore shrimp (heads-off pounds/acre)

| Type of model | Inshore | | Offshore | |
|---|---|---|---|---|
| | brown | white | brown | white |
| Non-lagged | 1.60 | 1.44 | 0.90 | 1.23 |
| Lagged | 1.25 | 0.86 | – | – |

lb/acre ≈ 1.12 kg/ha.

Our simplest estimates of the model excluded the lagged catch value. Separate regressions were performed for brown and white, and for inshore and offshore shrimp. There was no consideration of possible changes in the structure of the equations over time. We fit equation (4) for both the lagged and non-lagged case to estimate the values of the parameters $\beta_1$ and $\beta_2$. All the parameters and equations were significant with $R^2$ ranging from 0.67 to 0.90.

The marginal productivity of marsh based on equation (4) is then given by:

$$\mathrm{MP}_m = (\beta_1 + \beta_2 M) M/W \tag{5}$$

For inshore areas, the sample means of $M$ and $W$ for brown shrimp were 1426 days and 18 249 ha, respectively. The marginal productivity estimates based on these data are summarized in Table 1.

Unfortunately, there are not sufficient data to estimate the model employed above for menhaden, oysters, or crabs. For menhaden, we assumed that the marginal and average products of estuarine area (marsh plus open water) are equal for menhaden harvest. The entire 1983 Gulf Coast catch was 2036 million lb. Using total Gulf Coast estuarine area of 14 million acres, this implies an average and marginal productivity of approximately 145 lb/acre. This estimate could be biased upward because marginal product is generally lower than average product. It could also be biased downward since we divide through by the sum of marsh plus open water area. If marsh is more productive of menhaden per acre than open water (as we suspect it is) then this procedure underestimates the per-acre production of marsh. In lieu of better data, however, we feel our current estimate is the best that can be done.

We used the results from a study of the blue crab productivity of western Florida salt marshes for estimating the marginal product of Louisiana wetlands for blue crab. Lynne et al. (1981) estimated a bioeconomic model for blue crab harvesting that is similar to the system outlined above. They estimated that the marginal productivity of the Florida salt marsh is 2.3 lb

TABLE 2

Acres of marsh open water and total recorded pounds of oyster meat harvested from 1948 to 1978

| State | (a) Area of marsh and open water (million) | (b) Recorded total harvest (million lb) |
|---|---|---|
| La. | 7.3 | 283.5 |
| Fla. | 3.0 | 78.1 |
| Texas | 2.7 | 66.1 |
| Miss. | 0.6 | 47.7 |
| Ala. | 0.4 | 33.2 |
| Total | 14.0 | 508.6 |

Source: Allen et al. (1984, p. 120).

of blue crab per acre per year. Although they do not specifically derive the average salt marsh productivity, one can use their figures to derive an average productivity of approximately 28 lb of blue crab per acre of salt marsh per year. In contrast, they estimate that the marginal productivity of effort is 214 lb per trap per year, while the average productivity of traps is 435 lb per year. This illustrates the inaccuracy of using average productivities to measure environmental values.

We would have liked to estimate a model for oysters similar to the one for shrimp above. Data were by area of oyster landing rather than area of catch. Without data on the origins of oyster harvest it was not possible to relate harvest to the associated level of the environmental variables (marsh area).

The results of several other studies were therefore used to estimate the marginal product of wetlands for oysters. Batie and Wilson (1978) estimated the oyster productivity of Virginia wetlands using an equation similar to (1) and found a marginal product of 1.66 lb/acre in the county with the greatest wetlands area in their sample. Aggregate data presented in Allen et al. (1984) and shown in Table 2 can be used to estimate average oyster productivity of estuarine areas. This table shows estuarine area (marsh plus open water) in acres and pounds of oyster meat harvested as reported between 1948 and 1978 in five Gulf Coast states. Using the total values, the annual recorded harvest per acre of marsh plus open water over the 30-year period was 1.21 lb. A more sophisticated estimate of marginal product is obtained by regressing harvest in Table 2 ($H$) on estuarine acreage ($A$) shown in that table. The resulting least-squares equation is:

$$\underset{}{H} = \underset{(2.3)}{2.03} + \underset{(3.4)}{35.60A} \qquad F = 35.44 \tag{6}$$

TABLE 3

Summary of estimated economic value of wetland productivity for commercial fishery harvest.

| (a) Species | (b) Basis | (c) Annual MP estimate (lb/acre) | (d) 1983 ex-vessel price [a] ($/acre) | (e) Value of annual MP ($/acre) |
|---|---|---|---|---|
| Shrimp | Marsh Area | | | |
| brown inshore | | 1.60 | $2.10 | $3.36 |
| white | | 1.44 | 2.10 | 3.02 |
| brown offshore | | 0.90 | 2.10 | 1.89 |
| white | | 1.23 | 2.10 | 2.58 |
| Menhaden | Marsh and open water area | 145.00 [b] | 0.04 | 5.80 |
| Oyster | Marsh and open water area | 6.00 | 1.34 | 8.04 |
| Blue crab | Salt marsh area | 2.30 | 0.29 | 0.67 |
| Total | | | | $25.36 |

[a] USDC (1983).

[b] Assuming MP-average product.

Since the harvest data are for a 30-year period, the annual marginal product of estuarine area implied by equation (6) is 1.19 lb of oyster meat per acre. This estimate is roughly equal to the cruder average product estimate above.

The data in Table 2 are only for recorded harvests collected by NMFS. This includes oysters used only for canning. Sack and counter oysters are unreported. Lindall et al. (1972, p. 139) found that the total oyster production is five times the reported harvest. We assume a canning oyster marginal product of 1.20 lb, so the total marginal product is estimated to be 6.00 lb per acre of estuarine area per year.

Table 3 summarizes the commercial fishery productivities of marsh and estuarine areas by the four primary species. Column (b) shows the environmental variable being evaluated. For example, shrimp productivity was estimated by using the total of fresh and salt marsh acreage. Column (c) shows our annual marginal productivity estimates. Column (d) shows the 1983 ex-vessel price per pound. We assumed all shrimp were valued at the same price per pound. Column (e) is derived from columns (c) and (d). It shows the economic value of the marginal acre, and is the product of columns (c) and (d). For example, the estimated economic value of the annual marginal productivity for brown inshore shrimp was $3.36 per acre of marsh. The sum of all these economic values is $25.36 per acre of marsh per year. Discounting at 8% gives a present value per acre of $317. Discounting at 3% gives a present value per acre of $845.

*Fur trapping value of wetlands.* Muskrat and nutria accounted for 78% of the value of Louisiana trapping in 1977 (Davis, 1978, p. 87). The muskrat is primarily a brackish-to-intermediate marsh species. In this environment, the muskrat yield is approximately 0.98 pelts per acre. Freshwater marshes, the primary habitat of nutria, yield 0.88 pelts per acre (Davis, 1978, p. 85).

There is some difficulty placing a price on these pelts. The recent increase in the value of the dollar has reduced prices to less than one-half their 1979–81 value. We used the 1980–81 price of $6 for eastern muskrat pelts and $7 for eastern nutria pelts as expected average prices and assumed these prices to be constant over time. Using the pelt productivity values above, the annual value of pelt productivity is $12.04 per acre. At a 8% discount, this generates a present value of $151 per acre; at 3%, a present value of $401 per acre.

*Recreation value of wetlands.* Wetlands have values as recreational areas. They are used for fishing, hunting, photographing, boating, etc. These are non-commercial uses of wetlands. A survey of recreational users was undertaken on various days over a 1-year period to determine willingness-to-pay to preserve wetlands for recreational purposes in Terrebonne Parish, Louisiana. The survey was designed to utilize the travel cost method of evaluating consumer surplus from use of a site, and the contingent valuation method. The details of the survey and data are contained in Costanza and Farber (1985b) and Farber and Costanza (1987). Here we summarize the method and results.

The sampling procedure consisted of placing self-addressed, stamped questionnaires on windshields of all vehicles parked in the morning at 27 boat launch facilities in Terrebonne Parish on various dates throughout the period July 1984 to June 1985. The sum of the average number of vehicles per day across all sites was 563.29 on weekends and 132.1 on weekdays.

A total of 7837 questionnaires were distributed, and 1126 were returned for a response rate of 14.4%. There were 6248 questionnaires distributed on weekends, with a 15.0% response rate; and 1589 on weekdays with a 11.7% response rate.

There was no explicit attempt to determine non-response bias. However, we thought that persons placing a higher value on these particular wetlands would be more likely to return questionnaires. This could mean that response rates may decline with distance from the site. In order to test this, we instituted a lottery with several prizes ($50, $30, and $20) on weekends in

February and March, 1985. The response rate for weekend samples increased from 18.5% in the pre-lottery period to 24.3% in the lottery period.

In order to implement the travel cost methodology, seven rings of 35-mile * increments in radii were constructed centered at Dulac, Louisiana. Each parish or county of Louisiana, Texas, Mississippi and Alabama was placed in one of the rings or in a rest-of-world (ROW) category. The localized use of these wetlands is apparent from the fact that 78% of the respondents came from ring 1, and 98% from rings 1 through 3. This localized use may make the travel cost methodology inadequate for determining willingness-to-pay.

We estimated the value of travel time by determining the total cost of travel time to the typical user group in the sample. The typical user group consisted of 2.72 persons, with one male head, 0.66 female spouses, 0.51 children, and 0.55 non-family members. The average wage rate of male respondents in the sample was $13.37. Using the fact that a previous study found southern female wages to be 70% those of males, the average spouse wage was assumed to be $9.36. The non-family member was assumed to be male. Children were assumed to have zero wages. Weighting these wages by the number of members yielded average total hourly wages of the typical user group of $26.90. The estimated annual WTP using this travel cost method was $3.9 million/year.

A second method was to use respondents' answers to the direct question regarding their willingness-to-pay for the preservation of the Terrebonne Parish wetlands (contingent valuation). In particular, persons were asked what they would be willing to pay annually for the sake of their entire household. The mean response in the sample was $103.48 per household annually.

The number of different households using these wetlands annually must be estimated indirectly. An independent survey of recreational saltwater fishermen (Bertrand, 1980) found that 110 373 licensed fishermen used the LaFourche/Terrebonne area at least once in the 1982–83 season. Since Terrebonne constitutes approximately one-half of the total LaFourche/Terrebonne wetlands (Costanza and Farber, 1985b), we assumed only 55 186 of these persons used the Terrebonne area. Multiplying this by the annual WTP per household ($103.48) gives an annual total WTP of $5.7 million. This is slightly above the travel cost annual estimates. However, since the direct question referred to the head of household's WTP and several family members may have licenses, this direct estimate is not too far from the travel cost estimate. For example, if we assume the 2.17 members of a typical

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mile ≈ 1.609 km.

family user unit are licensed, the $103.48 amounts to $47.69 per person. The resulting annual WTP is then $2.6 million, only slightly less than the travel cost method result.

Depending on assumptions made about population growth, the present value estimates range from $34.2 million in the no growth situation to $40.8 million if population grows at a 1.3% annual rate when the discount rate is 8%. Assuming the population growth case, this amounts to $46 per acre of estuarine area (890 000 acres) using a 8% discount rate, and $181 per acre using a 3% discount rate.

*Storm protection value of wetlands.* Wetlands provide storm protection for urbanized areas in several ways. The marshes create surface friction for both tidal surges and winds. They also aid in reducing the heat source of energy for the storms. The result is that marshes help reduce tidal surge levels and wind velocities of storms. A very important value of coastal marshes is therefore their value in protection against personal and property damage due to storms

Costanza and Farber (1985b) contains our analysis of the hurricane protection value of wetlands for protection from storm surge and wind damage. Farber (1987) contains a more detailed analysis of the value of coastal wetlands for protection from wind damage only. We used U.S. Army Corps of Engineering estimates of property damages resulting from four recent Gulf Coast hurricanes, and adjusted these to 1983 dollars using a construction price index. These damage estimates included both wind and flooding damages. Damages were given by county. By measuring the distance of the center of a county from the coast, and controlling for population size, hurricane strength, distance from the path of the hurricane, and probability of hurricane incidence, we estimated the increase in expected property damages resulting from being closer to the landfall of a hurricane.

If Terrebonne wetlands receded by one mile, expected damages in a four-parish area would increase by $5 752 816 annually. The loss of a 207-ft wide strip running the length of Terrebonne parish coastline would increase expected damages by $128.30 per acre of the coastal strip annually. Discounting these values by 8% yields a present value of the strip of $71.9 million; and a present value of 1 acre in the coastal strip of $1604. Assuming population grows at 1.3% annually, the present value of such a strip for storm protection is $1915 per acre, or a total of $85.9 million for the entire strip. This is a relevant value for consideration of projects designed to stop rapid, wide-area recession and coastal erosion. It is not really appropriate for evaluating projects designed to affect erosion in areas that are not direct coastal areas; unless, of course, these internal areas eventually have an impact on coastal recession.

Another way of looking at the protection value of Terrebonne wetlands is to determine how much higher expected property damages would be if they were permitted to continue receding at their present rates. Using recent recession rates (approximately 12 ft/year), the present discounted value (at 8%) of increased expected property damages lies between $2.1 and $3.1 million. This is relevant in determining whether to initiate projects directed toward stopping long-term recession.

*Energy analysis of wetland values*

The theoretical basis for energy analysis as an economic value estimation tool is discussed in Costanza and Farber (1985b) and Turner et al. (1988). The method looks at the total amount of energy captured by natural ecosystems as an estimate of their potential to do useful work for the economy. It yields an estimate of a comprehensive (in that it should include all possible useful outputs) upper bound on the economic value of the products of natural systems. It is an upper bound because not all of the work done by the system is necessarily useful to the economy.

*Gross primary production.*

The energy analysis methodology can be quite complicated (cf. Costanza, 1980, Costanza and Herendeen,1984; Costanza and Neill, 1984). For the purposes of this study we employ a simplified technique that is readily calculable (we discuss its shortcomings later). This technique uses the Gross Primary Production (GPP) of the whole ecosystem as an index of the solar energy captured by the system, and converts this energy value into dollars using a single dollar–energy conversion factor (described below). GPP is a measure of the solar energy that is used by the plants in the system to fix carbon into organic molecules. This 'primary' production is then used to power all the plants and animals in the system. The plants and animals in the system also moderate water flow, sedimentation and other variables. GPP for an ecosystem can be thought of as analogous to GNP for an economy: both are crude (but essential) measures of overall system performance that say nothing about the internal distribution of production and must therefore be used with caution. GPP and GNP measure the value of inputs (or outputs) of ecological and economic systems, respectively.

The energy analysis procedure can be summarized as follows:

(1) Determine by field measurements and laboratory experiments the GPP of the natural area in question, under the with and without project conditions.

(2) Convert this estimate (usually measured in grams of carbon fixed per time unit or the heat equivalent energy content of the carbon) to fossil fuel equivalents (FFE) by considering the fuel efficiency of each source.

(3) Convert the FFE value into dollars using an economy-wide ratio of economic value per unit of energy, usually the ratio of GNP to total-economy energy use (measured in FFE).

All three steps involve uncertainty. Below we discuss the steps in more detail, pointing out the potential sources of error.

*GPP measurement methods.* Gross Primary Production (GPP) in plants is usually measured by analysis of gas exchange. During photosynthesis plants take up carbon dioxide from the air and fix it into higher energy organic molecules. They also give off oxygen as a by-product. GPP is a measure of how much carbon plants take up (or conversely, how much oxygen they give off) and, therefore, how much high-energy organic matter they create. The measurement is usually done with an infrared gas analyzer that detects carbon dioxide concentration differences for terrestrial plants, or with oxygen meters for submerged aquatic plants. Like most field measurements, GPP measurement is much more complicated than the simple description given above. Hopkinson (1978) gives a discussion of the technique and some of the problems and inaccuracies. Despite problems, GPP remains a popular and useful index of overall ecosystem energy capture. Table 4 gives some examples of GPP estimates for Louisiana wetlands and aquatic habitats.

*Conversion to fossil fuel equivalents.* GPP estimates are frequently stated in grams of carbon or calories of plant biomass per unit area per unit time. The first step in converting this to a measure of equivalent economic value is to convert it to energy units more directly relevant as input to the economy, i.e.

TABLE 4

Gross primary production (GPP) estimates for Louisiana wetland and open water habitats.

| Habitat type | Gross primary production (cal $m^{-2}$ $year^{-1}$) |
|---|---|
| Salt marsh | 48 000 |
| Brackish marsh | 70 300 |
| Fresh marsh | 48 500 |
| Average marsh | 55 600 |
| Salt aquatic | 6 600 |
| Brackish aquatic | 5 100 |
| Fresh aquatic | 9 300 |
| Average aquatic | 7 000 |
| Coastal plankton | 3 600 |

All values are directly from Hopkinson (1978). cal, calorie (International Table) = 4.1868 J (def).

TABLE 5

Energy quality conversion factors (from Odum and Odum, 1976)

| Type of energy | (a) Calorie cost (calories of heat to make 1 FFE calorie) | (b) Fossil-fuel equivalents (FFE calories/per heat calorie) (1/column a) |
|---|---|---|
| Heat from sun's rays, uncollected | 10000 | 0.0001 |
| Sunlight | 2000 | 0.0005 |
| Gross plant production | 20 | 0.05 |
| Wood, collected | 2 | 0.05 |
| Coal and oil, delivered for use | 1 | 1.0 |
| Energy in elevated water | 0.33 | 3.0 |
| Electricity | 0.25 | 4.0 |

fossil fuels. Fossil fuels are a much more concentrated, higher quality form of energy than plant biomass. One way of seeing this is to consider how much extra energy is required to upgrade biomass to fossil fuel; for example, in a biogas process. Another way is to consider the relative number of calories of biomass that would have to be burned in a power plant to produce the same amount of electricity as a given quantity of oil. Both of these methods have been used to estimate the 'energy quality factor' of biomass relative to fossil fuel. An approximate average is 0.05 cal biomass/cal fossil fuel (Odum and Odum, 1976), indicating that unprocessed biomass, such as that measured by GPP, is about 20 times less concentrated than fossil fuel. Table 5 lists energy quality conversion factors for various energy forms.

*Conversion to economic value.* One can look at the overall ratio of the value of economic output to energy input in the economy as a crude way to convert plant production to an equivalent economic value. This step is certainly the most controversial, with critics arguing that energy consumption and economic value are not necessarily related (Huettner, 1982). But recent studies provide supportive evidence that total direct and indirect energy consumption (embodied energy) and dollar values are indeed highly correlated in the U.S. economy (Costanza, 1980, 1984; Cleveland et al., 1984; Costanza and Herendeen, 1984). We therefore use a conversion factor based on these studies to give a crude estimate of the economic value of ecosystem production from their GPP estimates converted to FFE.

*Habitat interdependence.* The GPP technique outlined above does not account for the interdependence between habitats or differences in productivity within the same habitat type. For example, all salt marsh is assumed to have the same GPP, regardless of what other habitats it is adjacent to or to the special conditions of the site. This is an approximation, but the level of current knowledge is such that it is the best available approximation.

*Louisiana wetland value estimates via energy analysis.* Table 6 lists the gross primary production estimates for relevant Louisiana wetlands and aquatic systems and their estimated economic value using the simplified energy analysis methods outlined above. The values range from \$47 acre$^{-1}$ year$^{-1}$ for open ocean (coastal plankton) to \$914 acre$^{-1}$ year$^{-1}$ for brackish marsh. The relevant numbers are the change in value from wetland systems to open water systems, since this is the major impact of coastal erosion. These are listed in column (c). They range from \$509 acre$^{-1}$ year$^{-1}$ (for fresh marsh to fresh aquatic conversion) to \$847 acre$^{-1}$ year$^{-1}$ for brackish marsh to brackish aquatic conversion, with an average of \$631 acre$^{-1}$ year$^{-1}$. The

TABLE 6

Gross primary production and EA-based economic value estimates for relevant Louisiana wetland and marine habitats

| Habitat type | Total energy captured measured by GPP [a] (kcal m$^{-2}$ year$^{-1}$) | Annual equivalent dollar value [b] (\$ acre$^{-1}$ year$^{-1}$) | Net marsh-aquatic change in annual value (\$ acre$^{-1}$ year$^{-1}$) | Present value (\$ acre$^{-1}$) assuming specified discount rate [d] | |
|---|---|---|---|---|---|
| | | | | 8% | 3% |
| Salt marsh | 48000 | 624 | | | |
| Salt aquatic | 6600 | 86 | 538 | 6700 | 18000 |
| Brackish marsh | 70300 | 914 | | | |
| Brackish aquatic | 5130 | 67 | 847 | 10602 | 28200 |
| Fresh marsh | 48500 | 630 | | | |
| Fresh aquatic | 9300 | 121 | 509 | 6400 | 17000 |
| Coastal plankton | 3600 | 47 | Average 631 | 7900 | 21000 |
| Spoil banks [c] | 13000 | 169 | | | |

[a] GPP is gross primary production. Values are from Hopkinson 1979.

[b] Based on conversion factors of 0.05 coal equivalent (CE) kcal/GPP kcal (Table 5), and 15000 CE kcal/1983 dollar (from Costanza, 1980) and 4047 m$^2$ acre$^{-1}$. The overall conversion factor from GPP (in kcal m$^{-2}$) to estimated economic value (in \$ acre$^{-1}$ year$^{-1}$) is therefore: $(0.05\times4047)/15000 = 0.013$.

[c] Estimated from values for upland systems.

[d] Rounded to nearest \$100.

present value of an infinite stream of these values using 8% and 3% discount rates are given in columns (d) and (e), respectively. These yield values for average marsh/aquatic conversions of \$7900/acre and \$21 000/acre, respectively.

*Summary wetland value estimates*

The estimation of the economic value of natural wetlands is a difficult and complex task, but it is essential to rational management. We have employed both willingness-to-pay (WTP) and energy analysis (EA) techniques to improve this estimate. Table 7 summarizes our findings.

One problem in communicating our findings is that individual elements of the overall estimates are of varying precision and yet they must be added to produce a total average value. In Table 7 we list totals for the WTP-based analysis. We have not included any estimate for option and existence values, which studies in other areas have shown to be potentially large components of the total. This indicates that our total WTP estimate is probably still very conservative.

Our 'best estimate' values are given as a range in Table 7. These are \$2429–6400 per acre (assuming an 8% discount rate) and \$8977–17 000 per acre (assuming a 3% discount rate). Which discount rate may be most appropriate under given conditions is discussed further on, as are methods for dealing effectively with the current range of uncertainty in wetland valuation by using a assurance bonding fund system.

TABLE 7

Summary of wetland value estimates (1983 dollars)

| Method | Per-acre present value (\$) at specified discount rate | |
|---|---|---|
| | 8% | 3% |
| WTP based | | |
| commercial fishery | 317 | 846 |
| trapping | 151 | 401 |
| recreation | 46 | 181 |
| storm protection | 1915 | 7549 |
| Total | 2429 | 8977 |
| Option and existence values | ? | ? |
| EA-based GPP conversion | 6400–10 600 | 17 000–28 200 |
| 'Best estimate' | 2429– 6400 | 8977–17 000 |

## DISCUSSION OF CURRENT VALUATION RESULTS

This section discusses what we perceive as the potential estimation problems that we encountered in valuing wetlands and outlines future studies that would be most effective in improving the estimates.

### *WTP-based estimates*

First, the estimates of commercial values of wetlands may be overstated for some species since we used the average, rather than marginal, product of the marsh. Second, recreational value may be underestimated. The travel cost method works well if a site attracts persons from a wide range of distances. However, our evidence is that there is very intensive localized use of the Terrebonne wetlands. Since travel costs to local users are small, our estimate of their willingness-to-pay for access is also small. However, there may be no better techniques at the moment since directly asking persons their willingness-to-pay is plagued with many problems. Third, the value of wetlands for hurricane protection is important but our estimation methodology is crude. We feel it is very important to spend more effort in improving the accuracy of the flood damage benefits of wetlands. The estimation methodology we used probably overstates hurricane protection value since some of the hurricanes studied passed over coastal areas with significantly more upland area than those passing over Louisiana wetlands. Fourth, loss of wetland reduces habitats and food sources and reduces natural waste treatment capacity. We would need to know the WTP for loss of treatment capacity. Finally, we were unable to measure the value placed on wetlands by persons who do not directly use the wetland and/or who do not consume products produced by wetlands. For example, some people may be willing to pay to preserve wetlands in order to preserve the option of using them in the future. Also, people may obtain pleasure from knowing the wetlands exist; perhaps because of indirect pleasure they receive from such things as photographs, literature, etc. pertaining to those wetlands. These values may be considerable. These option and existence values would be good topics for future research. We attempted to estimate these through a statewide survey; however, the response rate was so low that we realized we would not have enough funds to complete the estimation.

### *EA-based estimates*

The EA estimate may be an upper bound on the total value of wetlands. In practice there is enough imprecision in the data and uncertainty in the methods to make it difficult to tell whether the actual numbers are over or underestimates of the true value. It was encouraging that the EA based estimate was higher than the total WTP based estimate by an amount that seemed reasonable, given the known omissions from the WTP estimate. But more research on the general relationship of energy use to economic activity, and the specific relationship of GPP to economic value are needed to improve the quality of this estimate, and to determine its role in the overall valuation procedure.

## DISCOUNTING

Often the present vs. future issue is thought to be objectively decided by discounting. But discounting at best only reflects the subjective valuation of the future to presently existing individual members of human society. Discounting is simply a numerical way to operationalize the value judgment that: (a) the near future is worth more than the distant future to the present generation of humans, and (b) beyond some point the worth of the future to the present generation of humans is negligible. Economists tend to treat discounting as rational, optimizing behavior based on people's inherent preferences for current over future consumption.

There is evidence, however, that discounting behavior may be symptomatic of a kind of semi-rational, sub-optimizing behavior known as a 'social trap'. A social trap is any situation in which the short-run, local reinforcements guiding individual behavior are inconsistent with the long-run, global best interest of the individual or society (Platt, 1973; Cross and Guyer, 1980; Costanza, 1987). We go through life making decisions about which path to take based largely on the 'road signs', the short-run, local, reinforcements that we perceive most directly. These short-run reinforcements can include monetary incentives, social acceptance or admonishment, and physical pleasure or pain. Problems arise, however, when the road signs are inaccurate or misleading. In these cases we can be trapped into following a path that is ultimately detrimental because of our reliance on the road signs. Discounting may allow individuals to give too little weight to the future (or other species, other groups or classes of humans, etc.) and thus helps to set the trap. Economists, while recognizing that individual behavior may not always lead to optimal social behavior, generally assume that discounting the future is an appropriate thing to do. The psychological evidence indicates, however, that humans have problems responding to reinforcements that are not immediate (in time and space), and can be led into disastrous situations *because* they discount too much.

It can therefore be argued that the discount rate used by the government for public policy decisions on common property resources (like wetlands) should be significantly lower than the rate used by individuals for private investment decisions. The government should have greater interest in the future than individuals currently in the market because continued social existence, stability, and harmony are public goods for which the government is responsible, and for which current individuals may not be willing to fully pay (Arrow, 1976).

Discounting future value by the rate of interest also provides a tight link between ecological destruction and macroeconomic policy. Any exploited species whose natural rate of population growth is less than the real rate of interest is under threat of extinction, even in the absence of common property problems. While Paul Voelker and the Federal Reserve probably do not worry about the effect of U.S. interest rate policy on deforestation in the Amazon or destruction of Louisiana wetlands, such links really do exist, and they probably should not.

In terms of our wetland valuation problem all this merely increases the uncertainty concerning the total present value of wetlands, because the appropriate discount rate is uncertain and it makes a big difference in the results. We have stated estimates for a range of discount rates (3%–8%) in order to demonstrate how much uncertainty is introduced by uncertainty in the discount rate, and have given arguments for why a lower discount rate may be more appropriate for wetland valuation decisions. Below we consider a method to deal with this uncertainty.

## DEALING WITH IMPRECISE VALUES USING AN ASSURANCE BONDING SYSTEM

One fundamental problem with our current methods of wetlands management is that those parties that damage and destroy wetlands are not charged for the true social cost of that damage or destruction. Two reasons often given for this are: (a) we don't know what to charge them and/or (b) charging them anything implies that wetlands can be sold and they will just pay the money and continue on their merry destructive ways. Therefore we cannot accept any charging scheme and must simply prevent all destruction of wetlands.

These ideas have guided wetlands management over the last several decades and the results have been abysmal. Wetland destruction has accelerated, not abated. Canal dredging and other hydrologically disruptive activities have resulted in a current land loss rate of over 100 $km^2$/year in coastal Louisiana (Craig et al., 1979; Gagliano et al., 1981; Scaife et al., 1983). To understand why it has not worked we must understand the structure of the short term reinforcements facing the various players in the game of wetlands management. The situation is a 'social trap' because the narrow, short term incentives of those damaging the wetlands are inconsistent with the long term good of the system (Costanza, 1987).

To turn this trap into a trade-off one should charge the parties responsible the full cost of the ultimate environmental damage, at the time the damage is done. To do this one needs to know the economic value to society of coastal marshes and the amount of marsh destroyed by each activity. Since there is much uncertainty involved in these estimates the 'worst case' costs should be assumed and the burden of proof that the damages are in fact less than the worst case shifted to the parties who benefited or stand to benefit (see Costanza and Perrings, 1990, for a more complete discussion of this approach).

For example, our study concluded that each acre of coastal wetlands in Louisiana has a present value to society of roughly \$2500–\$17000 per acre. This range of values was due to uncertainties in the valuation procedures and the discount rate. Increasing the accuracy of the valuation estimates is an expensive proposition, and one that would stress the research budget of the state.

To effectively eliminate this trap one could charge the parties responsible for marsh destruction (ie. oil companies for dredging access canals through wetlands or developers wishing to convert wetlands to other uses) the \$17 000 per acre worst-case cost. These fees would go into an assurance bond to be returned to the developers in the event that damages are less than the worst case, or to be used for mitigating environmental damages by purchasing marshland elsewhere, backfilling canals, freshwater diversion, etc. (Costanza and Perrings, 1990). It would allow managers much more flexibility than the current 'mitigation banking' schemes because it is not limited to simple acre for acre tradeoffs. The responsible parties could lower the fee or secure a return of the bond by proving that the damages are actually less than the worst case assumption (by funding independent studies) or by minimizing the amount of wetlands they damage in the process of accomplishing their goal (by directional drilling, immediate backfilling, etc.). In either case the cause of wetland conservation would be served without unduly hindering the search for oil and gas, by converting the trap to a trade-off. Wetland developers would have strong economic incentives to minimize their damage to wetlands, and to fund additional needed research into the functions of wetlands (in order to narrow the range of uncertainty in wetland valuation estimates). Money would be held in the bond to be paid back to developers if future research aimed at reducing the uncertainty of the valuation estimates proves that they were overcharged.

## SUMMARY AND CONCLUSIONS

Markets for ecological goods and services are far from perfect and we cannot rely on the free market to efficiently allocate these resources. The current system, which misallocates these resources, is better described as a social trap. Escaping from the trap involves turning it into an economic trade-off by making long run social costs, risks, and uncertainty incumbent on individuals and firms in the short run using fees and subsidies. Quantifying the proper level of fees and subsidies should be tied to models of the ecological impact of activities, but the burden of proof as to the magnitude of these damages should fall on the parties that stand to profit from them, not the general public. This addresses the distribution and incentive issues inherent in the problem in an equitable way. A flexible ecological cost charging and assurance bonding scheme can be designed that induces ecotechnological innovation by making it the most economically attractive option in the short run (as well as the long run). Implementation of such a scheme could go a long way toward allowing the development of a more ecological economics.

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