METHODOLOGICAL ISSUES ASSOCIATED WITH ESTIMATION OF THE ECONOMIC VALUE OF COASTAL WETLANDS IN IMPROVING WATER QUALITY

by

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November 1979

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ALTERNATIVE MANAGEMENT STRATEGIES FOR
VIRGINIA'S COASTAL WETLANDS

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PREFACE

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The results of the study are reported in a series of project papers under the general title: "Alternative Management Strategies for Virginia's Coastal Wetlands" with project paper titles as follows:

		Sea Grant Project #
1.	Alternative Management Strategies for Virginia's Coastal Wetlands: A Program of Study	not numbered
2.	Economic Implications of Environmental Legislation for Wetlands	VPI-SG-77-05
3.	Estimating the Economic Value of Natural Coastal Wetlands: A Cautionary Note	VPI-SG-77-06
4.	Existing Legal Framework for Management of Virginia Coastal Wetlands	VPI-SG-77-07
5.	The Development Value of Natural Coastal Wetlands: A Frameowrk for Analysis of Residential Values	VPI-SG-77-08
6.	Economic Values Attributable to Virginia's Coastal Wetlands as Inputs in Oyster Production	VPI-SG-77-04
7.	The Economics of Wetlands Preservation in Virginia	VPI-SG-79-07
8.	Estimating the Economic Value of Coastal Wetlands: Conceptual Issues and Research Needs	VPI-SG-79-08
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ABSTRACT

METHODOLOGICAL ISSUES ASSOCIATED WITH ESTIMATION OF THE ECONOMIC VALUE OF COASTAL WETLANDS IN IMPROVING WATER QUALITY

Coastal wetlands have been suggested as playing an important natural role in relation to the impact of nonpoint pollution on water quality. The traditional function attributed to wetlands involves long-term sediment accumulation. There is recent evidence to indicate that wetlands may have an important short-term buffering function, smoothing out the time pattern of pollutant loading to an estuary. The standard deterministic programming model which was developed to estimate the costs of agricultural nonpoint pollution control can be used in conjunction with the alternative cost technique to provide a maximum estimate of the value of the long-term sediment accumulation function. Data probably exists to make at least a rough estimate of the value of this function. A modification of the standard model to reflect the stochastic nature of nonpoint runoff will allow for estimation of the value of the short-term buffering function, if and when a sufficient data base is generated and the function is confirmed to be significant.

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INTRODUCTION

During the period of increasing environmental awareness of the last decade, legislation has been enacted providing for coastal wetlands regulatory programs in 14 states. The majority of these programs require permitting of those activities which may affect the provision of environmental services by wetlands [Rosenbaum, 1978]. Current permit decisions are being made with little knowledge of the costs of various activities in terms of reductions in social benefits associated with these services. To improve the efficiency of the allocation of wetlands to alternative uses, those responsible for permit decisions need information on the social value of the environmental services of wetlands [Batie and Wilson, 1978]. One of the environmental services commonly attributed to coastal wetlands is that of improving water quality.

In recent years it has become evident that nonpoint pollution control is the key to improving water quality in some areas [U.S. GAO]. Control strategies for agricultural nonpoint pollution generally call for modification of land use practices (e.g., tillage practices or crop rotations) which affect runoff rates and/or measures (e.g., buffer strips or grassed waterways) which seek to reduce the pollutant load of the runoff before it reaches the receiving water. Wetlands, by virtue of their being at the land-water interface, may well play a role similar to that of these latter measures. If this contribution is of significant value, preservation (and even creation) of wetlands is a relevant strategy for nonpoint pollution control.

The outline of this report is as follows. First, a brief literature review on the various ways in which coastal wetlands have been suggested to affect water quality is provided. These include use of coastal wetlands for tertiary treatment (i.e., removal of nutrients from secondarily treated sewage effluent) on a widespread or long-term basis, tidal exchange of nutrients between coastal wetlands and estuaries, and assimilation of pollutants in nonpoint runoff. Both a long-term accumulation function and a short-term buffering function in relation to nonpoint pollution loading are considered.

Second, economic methodologies for valuing the effect of wetlands on water quality are discussed. The lack of sufficient technical data precludes application of these economic methodologies at the current time, however, suggestions are made as to appropriate areas for future research.

COASTAL WETLANDS AND WATER QUALITY

Tertiary Treatment

The productivity of coastal wetlands vegetation, as vegetation in general, is usually limited by the supply of some nutrient. The logic behind using coastal wetlands for tertiary treatment is that additional amounts of the limiting nutrient will be assimilated through increased vegetative growth or retention in the soil. This would reduce the amount of this nutrient that is released to the estuary, relative to case where sewage effluent is discharged directly to the estuary. Thus, the rate of eutrophication would be reduced.

A number of recent studies have been undertaken to assess the extent to which this is the case. Most of these studies have involved the application of sewage sludge as fertilizer to experimental marsh plots to determine how much of the nitrogen or phosphorus inputs are retained in the plants or the soil. The general conclusion to be drawn from the work of Marshall [1970], Valiela and Teal [1974], Sullivan and Daiber [1974], Broome et al. [1975], Gallagher [1975], Patrick and Delaune [1976], and Chalmers et al. [1976] is that addition of nitrogen results in increased marsh productivity, but the addition of phosphorus has no effect. In other words, nitrogen appears to be the limiting factor in marsh productivity, which is consistent with the finding by Pomeroy et al. [1969], that there is an abundant supply of available phosphate in salt marsh soils.

More important is the question of how much of the applied nutrients are retained permanently by marshes as opposed to being released to the estuary. The most comprehensive study of this aspect of the impact of sewage on salt marshes has been done in Massachusetts. Valiela, Teal, and Sass [1973] reported that their experimental marsh retained 80-94% of the nitrogen in the sludge fertilizer, although only 12% of the nitrogen added could be accounted for in the annual above-ground growth [Valiela, Teal, and Van Raalte, 1973]. They emphasize that their study was only a preliminary inquiry, and that they do not know what the long-term capacity of marsh for nutrient retention is.

In a similar study in Georgia, Chalmers *et al.* [1976] reported retention of only about 50% of the sludge nitrogen applied in the marsh plants and soils. They concluded that the difference between the Georgia and Massachusetts results may be a result of a significantly higher rate of sedimentation for New England marshes relative to those of Southeastern ones. From another loading experiment using two Chesapeake Bay tidal marshes in Maryland, Bender and Correll [1974] concluded that low marsh (i.e., marsh which is inundated regularly during the tidal cycle) had no measurable capacity to act as a permanent sink for nitrogen or phosphorus. Although high marsh (i.e., marsh which is inundated only at what may be lengthy intervals) had no capacity for assimilating nitrogen, it had some capacity for phosphorus. However, this capacity was used up in 45 days, and probably could not be reused for many years.

Thus, the long-term capacity of marsh for assimilation of nutrients from artificial sewage sludge loading remains an open question.

Other studies have been done of coastal wetlands which are located downstream from sewage outfalls. Grant and Patrick [1970], in a study of Tinicum Marsh which receives sewage outfall from three plants on Darby Creek outside Philadelphia, Pennsylvania, suggest that this brackish tidal marsh plays an important role in reducing concentrations of BOD, nitrates, ammonia, and phosphates as the creek flows through the marsh. However, the irregular pattern of flow and the limited period of sampling (August, September, and October) preclude any exact measures or strong conclusions. Mattson *et al.* [1975] indicates that the brackish Hackensack Marshes in New Jersey are very efficient at taking up nitrogen from water flowing through them.

However, Wetzel et al. [1977] studied an historically nutrient enriched marsh ecosystem in a subestuary of the lower Chesapeake Bay. This brackish marsh received secondarily treated sewage input from one of the two main creek branches for about 19 years. The authors concluded that the marsh system did not effectively assimilate nitrogen and phosphorus, though it did affect the form of nitrogen, converting nitrate into ammonia. They also suggest, as do Henile and Flemer [1976], that long-term nutrient loading by treated sewage may have changed the marsh ecosystem to such an extent that "normal" relationships are no longer operating.

Thus, the long-term provision of tertiary treatment by coastal wetlands located downstream from sewage outfalls is questionable.

Tidal Exchange

The process of tidal exchange of water between coastal wetlands and estuaries has been the focus of much study. One of the research questions has been whether coastal marshes are significant net importers of nutrients; i.e., whether coastal marshes import more nutrients from the incoming tide than they export to the outgoing tide. In such case, coastal marshes could again be considered to be reducing the rate of eutrophication. With regard to nutrient transport in the process of tidal exchange, Aurand and Darber [1973] found a net import of inorganic nitrogen from the estuary to a Delaware marsh. Axelrad, Moore and Bender [1976] studied annual nitrogen, phosphorus and carbon budgets for two Virginia salt marshes. They found a strong overall net export of nitrogen from marsh to estuary, though there tends to be a net import of particulate nitrogen during spring and summer and a dominating net export of dissolved organic nitrogen. The data suggest that marsh imports particulate phosphorus and exports dissolved inorganic and organic phosphorus, with a net phosphorus loss from estuary to marsh. They conclude that the contribution of marsh to water quality is in mineralizing imported particulate organic nitrogen and phosphorus and exporting these nutrients in a dissolved form. The dissolved forms can then be more easily assimilated by estuarine organisms. Stevenson et al. [1976] similarly found in their study of two Chesapeake Bay marshes in Maryland that there is a net export of dissolved nitrogen and phosphorus from the marsh to the estuary, though inorganic nitrogen is taken up in May, June, and July. They concluded that nutrient removal capacity on an annual basis is very limited in brackish marshes, though other studies suggest a strong capacity in high salinity areas.

Here too, then, there remains an open question as to the effect of marsh processes on water quality.

Nonpoint Pollution

Long-term accumulation function

The basic function in relation to nonpoint pollution runoff attributed historically to coastal marsh has been that of long-term sediment trapping. As runoff moves through wetlands, the reduction in velocity and presence of vegetation causes sediment to fall out of suspension. Boto and Patrick [1978], in an excellent review of literature related to the role of wetlands in the removal of suspended sediments, note that "sediment accretion rates...vary widely according to location depending on the input from river-borne sediment, elevation, tidal regime, storm surges, and in some instances, man-induced effects."

Sediment accretion has generally been measured in terms of average annual rates. To the extent that sediment is deposited permanently in coastal wetlands, there may be reduction in costs associated with turbidity (e.g., damages to aquatic life) or with deposition of sediment in estuaries (e.g., for dredging). To the extent that nutrients are adsorbed to sediment which is deposited in wetlands, the rate of eutrophication may also be reduced. To the extent that toxic materials such as pesticides and heavy metals are adsorbed to sediment which is deposited in wetlands, decomposition or burial of the toxic substances may occur.

Boto and Patrick [1978] note that "there is a large amount of data available on the affinity of nutrients and various toxic materials for sediments" and discuss some recent references since a review by Oschwald [1972]. Input rates of nitrogen and phosphorus to a streamside marsh were estimated to be 210 kg/ha/yr and 16.5 kg/ha/yr respectively by DeLaune *et al.* [1978]. However, Boto and Patrick [1978] acknowledge that "much of this discussion (of the effect of sedimentation in coastal wetlands on water quality) is speculative in that few actual studies of these effects in practical situations are available." There remains too the possibility that a hurricane-related storm, may completely scour a tidal creek, depositing in the estuary sediment which has accumulated over many years in the coastal marsh system [Correll, 1979].

Short-term buffering function

It is surprising to the authors that so little attention has been paid to the fact that sediment accretion in coastal wetlands does not proceed at a steady rate over the period of a year, but at relative high rates for short periods of time associated with storm events. Consideration of only annual average rates ignores potential water quality problems related to extreme short-term rates of pollutant loading, even where annual average rates do not appear to be a problem.

Monitoring of runoff from Maryland's Rhode River watershed for only three years indicates that short-term variation in loading rates on a storm event, seasonal, or even annual basis can be tremendous. Over a three-year period (1974-76), annual and summer total phosphorus discharge varied up to 68% and 90% from the means. Data on two dif-- ferent weeks with storm events in the spring of 1975 provides even more startling evidence. The amounts of total phosphorus in the two runoff periods were 86.32 and 4.26 kilograms [Correll et al., 1977].

Two examples will suffice to illustrate why such variability may be critical from the standpoint of water quality. One, long-term sediment loading at a moderate pace may not affect oyster yields, whereas an extreme load related to a storm even may reduce them significantly. Two, longterm nutrient loading at a moderate pace may be assimilated by a re-ceiving water body, whereas a heavy load related to an abnormally wet summer may cause the receiving water body to turn anaerobic. The consequence of anerobic waters can be widespread damage to the biota and unaesthetic recreational conditions.

Recognition that the variances associated with mean rates of pollutant loading, on a storm event, seasonal, or annual basis, may be relevant from the standpoint of water pollution damages suggests consideration of whether and how wetlands affect the variance as well as the mean of pollutant loadings. Discussion with Dr. David Correll of the Chesapeake Bay Center for Environmental Studies provided the basis for formulating an hypothesis as to how coastal wetlands may reduce the variance of pollutant loading to estuaries by smoothing out the time pattern of pollutant loading. Such a buffering function can be characterized graphically as in Figure 1, where the rate of pollutant loading (in lbs. per hour for example) associated with a storm or a wet month, with and without a wetlands system, is represented over time. Thus, the level of loading for any period of time is equal to the area under the curve. Although the mean level of pollutant loading over the two equal time periods, T_0T_1 and T_1T_2, are approximately equal for the two cases (i.e., the area under the curves from T_0 to T_2 are approximately equal), the variances differ markedly. If pollutant loading above the level represented by the hatched area for the period of length T_0T_1 is a potential receiving water quality problem, the preservation (or creation) of wetlands may be of significant value, even if the mean level pollutant loading is not reduced.

There is some tentative empirical evidence to support this hypothesis that wetlands smooth out or buffer the time pattern of pollutant loading, thereby reducing the variance of pollutant loading levels. Phosphorus loading into a tidal wetlands system from land runoff and tidal exchange of phosphorus was monitored over a 13-month period [Correll, 1979]. Excluding a three-month period of severe drought, the mean monthly phosphorus load from land runoff to the tidal wetlands system was not significantly different from the mean monthly net export of phosphorus from the tidal wetlands system to the estuary. However, the variance associated with the mean load to the wetlands was significantly greater than the variance associated with the mean net export from the wetlands to the estuary. It appears reasonable to assume that

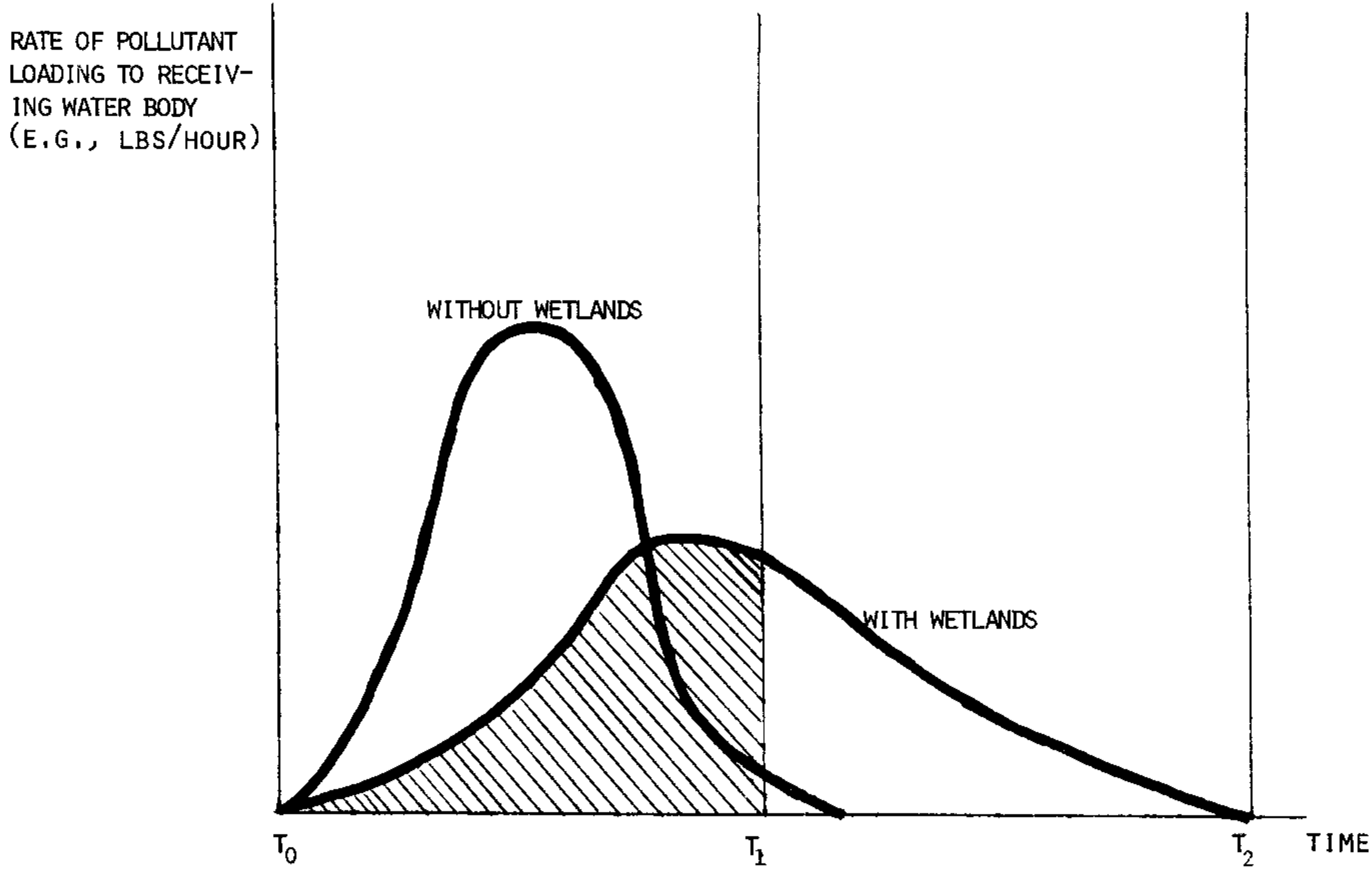

FIGURE 1. BUFFERING EFFECT OF WETLANDS ON POLLUTANT LOADING

development of the wetlands would result in the whole monthly phosphorus loading from land runoff to the wetlands being exported from the wetlands to the estuary in the same month; that is, the distribution associated with loading to the wetlands would apply to the net export from the wetlands to the estuary. Assuming normal distributions, the distributions associated with the mean monthly net export of phosphorus from the wetlands to the estuary, with and without wetlands, can be represented as in Figure 2. As can be seen, development of the wetlands could be expected to increase the probability of exceeding a monthly net export of phosphorus to the estuary of 600 kilograms (a damage threshold perhaps) from approximately .02 (ABC) to .13 (ADE).

Some additional evidence in support of this hypothesis of a short-term buffering function is found in Boto and Patrick's [1978] review. They note, in regard to the findings of Richard [1978], that "[i]t is of interest that the seasonal effects were much less pronounced for the established Spartina colony, perhaps indicative of a buffering effect of the vegetation." They also note in regard to the findings of Harrison and Bloom [1977], that "in years with fewer than average storms, less accretion was observed...[which] may point to the effect of vegetation in buffering storm surges and allowing accretion rather than erosion of sediment."

Thus, to the extent that extreme short-term rates of pollutant loading pose a potential water quality problem, coastal wetlands may have two valuable functions in relation to nonpoint pollution control, long-term accumulation and short-term buffering.

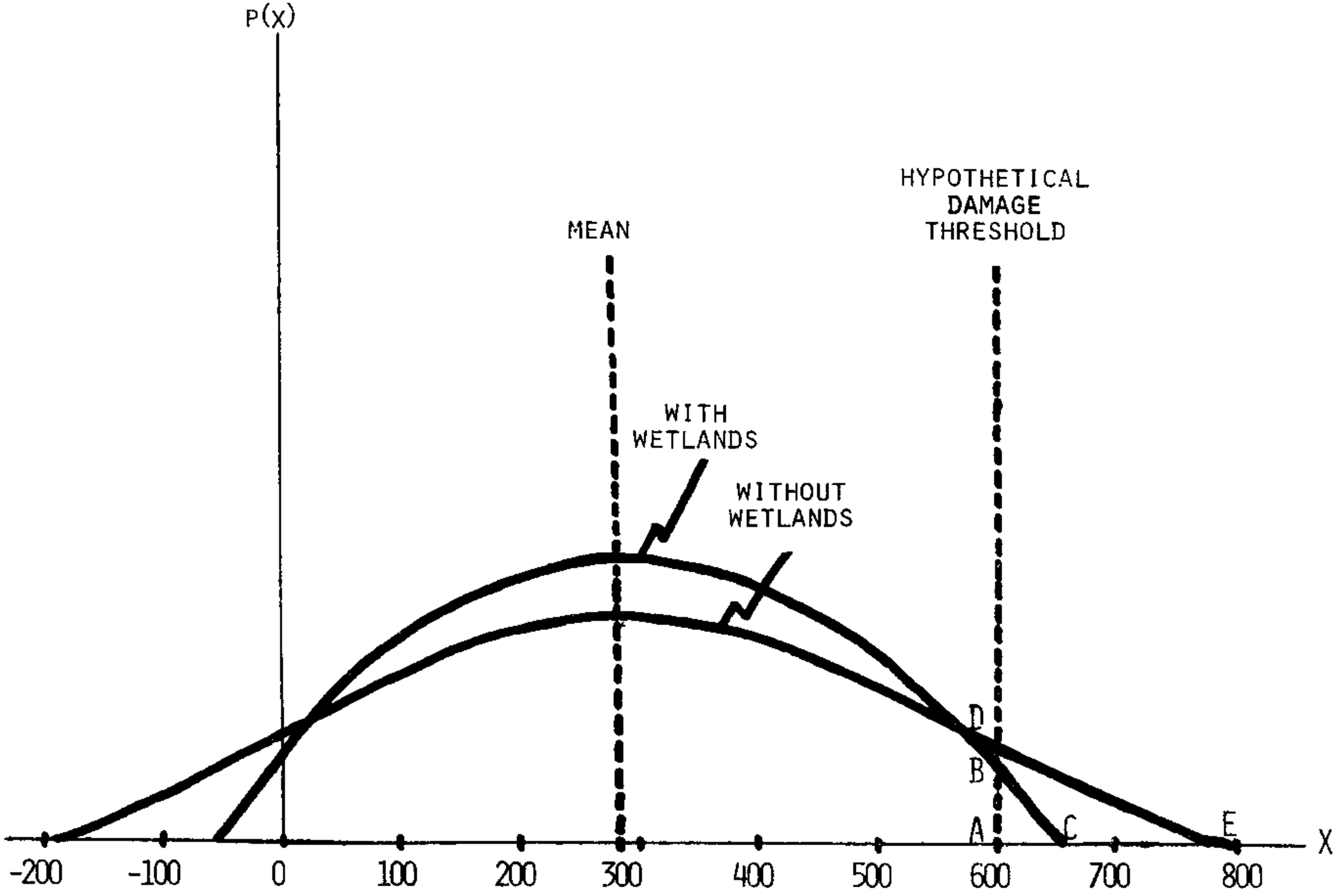

FIGURE 2. EFFECT OF WETLANDS ON THE DISTRIBUTION OF MONTHLY NET EXPORTS OF PHOSPHORUS (IN LBS.) FROM THE WETLANDS TO THE ESTUARY

ECONOMIC METHODOLOGIES

When certain conditions hold, the alternative cost technique provides a means of estimating a maximum bound on the value of coastal wetlands' roles in improving water quality [Batie and Shabman, 1977]. The logic is that wetlands' functions in regard to water quality improvement can be valued at no more than the cost of providing the <u>same</u> amount of improvement by the next most costly alternative method. The assumption implicit in this logic is that society is willing to pay at least the cost associated with this alternative method for this amount of improvement.

Tertiary Treatment

Despite the fact, indicated previously, that the long-term capacity of coastal wetlands to assimilate heavy nutrient loadings is questionable, some authors have applied various methodologies in an attempt to value this purported service of wetlands. Gossselink, Odum and Pope [1974] converted the Valiela *et al.* [1973] sewage effluent loading results for phosphorus into an annual dollar value of $480 per acre by applying an alternative cost of approximately $1.20 per pound for phosphorus removal by conventional methods. Bender and Correll [1974] similarly converted their results to an annual dollar value of $158 per acre. Shabman and Batie [1978] have criticized in detail the application of estimates based on the valuation methodology suggested by Gosselink *et al.* to coastal wetlands in general.

There are four basic criticisms discussed by Shabman and Batie. First, it appears that the degree of assimilative capacity of marsh varies greatly, making any valuation site specific. Second, only those wetlands plots that are actually used for nutrient assimilation have any value for that purpose. The costs of transporting sewage sludge from its source to a marsh plot are not included and would probably be prohibitive except for nearby marsh. Third, it is not clear that $1.20 per pound for phosphorus removal is necessarily associated with the least-cost alternative. Finally, and most importantly, the implicit assumption that a social willingness to pay for tertiary waste treatment exists may not be valid. In general, society may not value the incremental improvement in water quality from tertiary treatment at anywhere near the alternative cost of providing it. To the authors' knowledge, the alternative cost technique is yet to be appropriately applied in valuing this function.

Tidal Exchange

Since the significance of coastal wetlands contribution to water quality improvement in the process of tidal exchange is not clearly documented, methodologies for valuing it will not be considered.

Nonpoint Pollution

Long-term accumulation function

For rural watersheds, the least cost alternative method of reducing pollutant loading to estuaries would appear to be adjustment of agricultural land use practices. If society is willing to pay to maintain existing levels of water quality, the long-term accumulation function of coastal wetlands can be valued by valid employment of the alternative cost technique. If coastal wetlands were developed, the least-cost alternative method of providing the same service of reducing long-term pollutant loading to estuaries would involve adjustment of agricultural land use practices so as to reduce pollutant loading in storm runoff. The cost can then be estimated in terms of reductions in net returns to farmers which are associated with adjustments in land use practices designed to reduce annual average sediment (or nutrient) loading from agricultural land.

A fairly standard methodology has developed for estimating the cost of reducing annual average sediment loss from agricultural land. In its most common form, the methodology that has been previously employed in estimating the cost of nonpoint pollution involves construction of a linear programming model of a single farm enterprise or of a large agricultural watershed which is assumed to operate as a single decision making unit [Casler, 1975; Gossett and Whittlesey, 1976; Horner, 1977; Jacobs and Timmons, 1974; Onishi and Swanson, 1974; U.S.D.A., 1976; Williams and Hann, Jr., 1978]. Normally, the objective of the programming model is the maximization of annual net revenue, i.e., maximization of the present value of returns to land and/or management. The land use activities of the models generally include crop rotations, tillage practices, and conservation practices such as contouring or terracing. The set is usually limited to those practices for which the Universal Soil Loss Equation can be used to provide estimates of average annual soil losses. Measures to intercept runoff between the field and the receiving water could be included as activities in the model, though this is not usually done. The constraints of the model, in addition to the usual resource constraints, include limiting average annual sediment loss associated with the land use activities. To estimate the cost of control, the sediment loss constraints are tightened from a base solution where they are not effective. The reduction in the objective function value associated with the adjustment in land use activities necessary to meet the new constraint represents an estimate of the cost of controlling nonpoint pollution from the base level to the constrained level. In addition, the cost-efficient practices are evident from new activities which come into solution.

The primary limitation of this approach for estimating the cost of nonpoint pollution control is the lack of a defined linkage to ambient water quality (and ultimately water quality benefits). It is the quantity of sediment that reaches receiving water bodies which is of real concern, not what is lost from the field. General assumptions about delivery ratios (i.e., the proportion of sediment lost from the field which is actually delivered to the estuary) are often used, although they frequently lack empirical basis. In addition, the research focus on soil loss involves an implicit assumption that either sediment is the pollutant of sole concern or losses of other pollutants such as nutrients and pesticides are directly and significantly related to loss of sediment. This assumption may not be true. Even considering such qualifications, however, if longterm pollutant loading over a period of many years is the only potential water quality problem, the methodology outlined above can be appropriate for estimating the cost of nonpoint pollution control by adjustment of agricultural land use practices.

A maximum bound on the value of coastal wetlands' long-term accumulation function can be valued within the framework of this methodology in the following manner. The effect of changing wetland acreage on water quality may be represented in the above model by adjusting the bounds of the annual loading constraints. Assume that with existing land use activities and coastal wetlands acreage, the legislated water quality goal for an estuary is just being achieved. Now, suppose the coastal wetlands adjacent to the tidal creek are to be developed to provide waterfront lots. Water quality in the estuary would be diminished. To compensate for the loss of the wetlands' effect (i.e., to maintain the water quality level), changes in land use activities must be undertaken. In terms of the linear programming model, if water quality standards are to be maintained, the bounds of the pollutant loading constraints must be tightened, forcing adjustment in the land use activities to substitute for the eliminated average annual effect of the wetlands on water quality. The reduction in the objective function value associated with this tightening of constraints represents an estimate of the value of the coastal marsh in controlling nonpoint pollution. As indicated above, this estimate of value is valid as long as the conditions necessary for employing the alternative cost technique are satisfied. Existing estimates of annual average sediment loss from agricultural land and sediment accretion rates for coastal wetlands would probably allow rough estimates of the value of this long-term function to be made.

Short-term buffering function

If there is a potential water quality problem associated with extreme rates of short-term pollutant loading, the buffering effect of wetlands may be of significant value. However, the linear programming methodology presented above is no longer appropriate once the stochastic nature of nonpoint runoff is acknowledged as important.

The most realistic formulation of the stochastic elements would appear to be captured in a chance-constrained programming problem with the land use activities having stochastic loading coefficients (a_{ij}'s). A representative constraint on the loading of the j^{th} pollutant would then be set up in the following way:

$$[P(a_{1j}X_1 + a_{2j}X_2 + \ldots + a_{nj}X_n) \leq b_j] \geq p$$

where:

X_i = the level of the i^{th} activity,

a_{ij} = the loading of the j^{th} pollutant associated with one unit of the i^{th} activity (stochastic) for some time period, t,

b_j = the constraint bound for the j^{th} pollutant for some time period, t, and

p = the critical probability level.

Thus, this representative constraint requires that the probability of loading of the j^{th} pollutant being less than b be greater than or equal to p. For example, the value of p could be set at .95. In that case, the constraint requires the probability of loading of the j^{th} pollutant being greater than b to be less than or equal to (1 - p), which would be .05. An implicit assumption here is that society desires, and is willing to pay, to protect itself from violating the ambient water quality standard except in the case of the extreme event--which occurs only 5% of the time. This is the approach taken in public investment for flood control, where structures are designed to control up to, say, the 100-year flood.[1]

The procedure for estimating the cost of nonpoint pollution control within the framework of this model is as follows. A receiving water quality goal in the form of a b_j is selected, perhaps related to a pollution damage "threshold." The probability of nonviolation (i.e., of the j^{th} pollutant loading not exceeding b_j) associated with unconstrained choice of land use activities, say p^0, is determined. Then p is increased to a socially desirable level, p^1. As before, the objective function reduction associated with the adjustment in land use activities necessary to increase p from p^0 to p^1 is an estimate of the cost of this level of nonpoint pollution control. Alternatively, one could select p to begin with and estimate the cost of reducing the level of pollutant loading b_j which would have p probability of being exceeded.[2]

The use of the stochastic methodology outlined above to evaluate this function of wetlands relative to the cost of conventional agricultural land use practices proceeds similarly to that of the deterministic one and can be illustrated as follows. A wetlands activity would be specified as an activity with a stochastic pollutant loading coefficient with a negative mean. Suppose then that, with profit maximizing land use activities and existing wetlands acreage in a watershed, the water quality standard for pollutant loading to a receiving water body is barely being achieved. For example, suppose the standard requires that a certain pollutant loading rate can be expected to be exceeded no more than five percent of the time. Also assume that all the wetlands are to be developed to provide waterfront lots with access to a lake or estuary. The wetlands activity would be constrained to a zero level. Presumably, the variance in pollutant loading rates to the receiving water body, and perhaps the mean too, would be increased and the standard violated. With the model constrained to meet the water quality standard, the reduction in the objective function associated with the necessary adjustment in land use activities would represent an estimate of the value of this wetlands function.[3]

It is appropriate to ask whether computational difficulty and data limitations do not make such a stochastic formulaton inoperational. In regard to the former, Chen [1973] used such a model to analyze the cost of livestock feed rations with a chance-constraint on protein content. The stochastic constraint required the probability of a certan protein content being met or exceeded to be greater than or equal to some level. The solution procedure employed involved conversion of the chance-constrained program into a quadratic program. The necessary conditions for such a conversion are (1) there be only one chance-constraint, and (2) the joint probability distribution of the stochastic coefficients ($A_j'X$) be normal [Anderson <u>et al.</u>, 1977]. This joint distribution is likely to have a significant positive skew in the problem at hand. If this is the case, or if multiple chance constraints are imperative, it may be necessary to convert the chance-constrained program to an almost equivalent deterministic program for solution, perhaps in one of the ways that Chen outlines. However, another approach is available.

The fact that the stochastic loading coefficients can be expected to have high positive correlation across activities and perhaps across pollutants may allow a deterministic formulation analogous to the flood control approach. If all loading coefficients can be assumed to be strict positive functions of the level of rainfall[4] (i.e., if it can be assumed that a year, season, or storm with greater rainfall than another year, season, or storm will have relatively higher loading of each pollutant by each land use activity), a "design" event could be used to specify fixed loading coefficients. These loading coefficients would represent the level of loading for the "design" year, season, or storm that would be expected to be exceeded only one in 20 periods for example. The probability of the deterministic loading constraint being violated in actuality would be .05 as before. The advantage of this deterministic formulation over the stochastic formulation in terms of computational difficulty may outweigh the disadvantage of such an assumption with regard to the relationship among loading coefficients. As is usually the case, the choice between models depends on the tradeoff between realism and tractability.

It is appropriate to ask, in addition, whether data limitations do or do not make both the stochastic and the equivalent deterministic formulations inoperational. Monitoring of nonpoint runoff for several years at many sites is generating a large time series and cross-sectional data base. The development of this data, and the existence of considerable historical rainfall data, suggests that the potential for developing probability distributions for loading coefficients for a year, season, or a storm will soon exist. At this time it may be that probability distributions for pollutant loading associated with only broad land uses (such as row crop, grass crop, pasture, and forest) employing conventional practices can be constructed. In such case, the cost of reducing pollutant loading by adjusting land use among these classes would represent an upper bound estimate of control costs, since there may actually be adjustments within classes which would allow the same reduction in pollutant loading at lower cost. However, intensive effort currently being made to quantitatively measure the effectiveness of adjustments in land use practices for controlling agricultural non-point pollution should allow more detailed specification in the near future.

The real missing link is data regarding the effect of wetlands on the variance associated with mean levels of pollutant loading on a storm event, seasonal, or even annual basis. This lack of data would prohibit application of this methodology presently, even if data existed for conventional agricultural land use practices. Hopefully, further monitoring of wetlands processes will serve to provide a more extensive data base.

CONCLUSIONS

If coastal wetlands management is to improve the efficiency in the allocation of wetlands to alternative uses, information on the social value of environmental services of wetlands is required. Coastal wetlands are widely reputed to contribute significantly to the improvement of water quality. Their ability to provide tertiary treatment of sewage effluent has been documented, though long-term use is questionable. Their role in the process of tidal exchange does not appear to be significant. Their role in relation to nonpoint pollution may be of significance both on a long-term and short-term basis. The long-term accumulation of sediment and material adsorbed to sediment can be measured on an average annual basis. The short-term buffering effect of coastal wetlands may reduce the variance in levels of pollutant loading to an estuary on a monthly, seasonal, or annual basis. The alternative cost technique provides a framework for placing maximum bounds on the values of these contributions of coastal wetlands to improving water quality.

The tertiary treatment function can be valued by consideration of the cost of nutrient removal by conventional methods, though the costs of transporting sewage effluent must be considered. The long-term accumulation function in relation to nonpoint pollution can be valued by use of the standard linear programming methodology developed for estimating the cost of nonpoint pollution control. The short-term buffering function in relation to nonpoint pollution can be valued by use of a chance-constrained programming methodology which incorporates the stochastic nature of nonpoint pollution, or an almost equivalent deterministic approach. In each case the validity of the value estimates depends on the conditions for use of the alternative cost technique holding.

The lack of appropriate technical and physical data prohibits rigorous application of the programming methodologies to the nonpoint pollution control functions at the present time. Though coastal wetlands management has been a reality for over a decade, physical and biological scientists have directed relatively little attention toward analysis of the role of coastal wetlands in relation to nonpoint pollution. Perhaps this has been due to the belated recognition of nonpoint pollution as the controlling factor with regard to water quality in some areas. At the same time, this lack of attention underscores the failure of economists to communicate to physical and biological scientists what kind of technical data is necessary to do economic analysis that will be of use to policy-makers.

The stochastic nature of nonpoint runoff has general implications as well for the necessary interaction of economists with biological and physical scientists. First, the nature of potential receiving water quality problems must be identified more specifically. Second, nonpoint pollution control measures must be evaluated for their ability to reduce the variance associated with pollutant loading rates, as well as the mean, if excessive short-term rates of loading constitute a potential water quality problem.

FOOTNOTES

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[1]The standard linear programming models using average annual loading coefficients implicitly set p at .5 if the joint probability distribution of the loading coefficients are normal, and somewhat higher to the extent that the joint probability distribution is positively skewed. Thus, the expected frequency of actual violations is unknown.

[2]Hochman, et al. [1977], developed a model to determine the probability of shellfish contamination from nonpoint dairy farm runoff which would maximize joint profits of the two industries. Lowering the probability of contamination to .02, for example, required farms to install drainage ponds sufficient to hold the 50-year storm, since contamination was considered an all-or-nothing phenomena. Calculus or search methods were required for solution. The programming methodology proposed in this paper provides a more flexible empirical framework, allowing consideration of a variety of land use activity adjustments efficiently and adjustment of the level of pollutant loading as well as the probability of violation. As such, it is better-suited for the problems addressed in this paper.

[3]This all-or-nothing scenario will give only a total value for the wetland system (and thus average value per acre). Ideally, one would reduce the wetlands activity by one acre at a time to trace out what would be expected to be a downward slope marginal value curve. However, data on the wetlands effect for various acreage levels at the

same site is unlikely to be available and transfer of data across sites is unlikely to be valid.

[4]The use of some kind of an index, based on rainfall intensity as well as amount, and antecedent soil moisture conditions, would be desirable at a conceptual level, though the complexity of designing one may make it impractical for empirical work.

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