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Workshop on Marine Pollution and Environmental Damage Assessment

Co-sponsored by

Association of Environmental and Resource Economists

U.S. Environmental Protection Agency

and

National Oceanographic and Atmospheric Administration

Narragansett, Rhode Island
June 5-6, 1986

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Workshop on Marine Pollution and Environmental Damage Assessment
Preliminary Agenda

Thursday, June 5, 1986

8:00 - 8:30 am Sign-in Period

8:30 - 8:45 am AERE Workshop Program,
V.Kerry Smith, Vanderbilt University

8:45 - 9:00 am Introduction to Workshop,
Glen D. Anderson, U.S. EPA

9:00 -12:00 am SESSION I. Economic Valuation in a Policy Context

Session Chair: Glen D. Anderson, U.S. EPA

Compensation for Natural Resource Damages:
An Emerging Federal Framework
Roger C. Dower and Paul F. Scodari,
Environmental Law Institute

Measuring Damages to Coastal amd Marine Natural Resources from Oil and Hazardous Substance Spills: Application of an Integrated Ocean Systems/Economic Model
Thomas A. Grigalunas and James J. Opaluch
University of Rhode Island

Economic and Environmental Conflicts in OCS Oil and Gas Leasing: An Analysis of National VS Regional Benefits and Costs
Philip E. Sorensen
Florida State University

Break
Open Discussion

1:30 - 4:30 Session II. Economic Methodologies and Ecological Constraints: Case Studies of Marine Pollution

Session Chair: Steven F. Edwards, Woods Hole Oceanographic Institute

Regulation of Marine Contamination Under Environmental Uncertainty: Shellfish Contamination in California
Erik Lichtenberg, Western Consortium of Health Professionals
David Zilberman, University of Calif., Berkeley

Mitigating Damages from Coastal Wetlands Development: Policy, Economics and Financing
Leonard A. Shabman and Sandra S. Batie,
Virginia Polytechnic Institute and State University

Measuring the Economic Damages Associated with Terrestrial Pollution of Marine Ecosystems
James R. Kahn, SUNY-Binghampton

Break
Open Discussion

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Friday June 6, 1986

8:30 - 12:00 SESSION III. Conceptual Nees for Marine Pollution Policy

Session Chair: George Parsons, U.S. EPA

Economic Valuation of Wildlife:
Does Existance Value Exist?
Ronda K. Hageman, San Diego State University

Risk Sharing and Liability in the Control
of Stochastic Externalities
Kathleen Segerson, University of Wisconsin-Madison

Paper Title to Be Announced
Michael Hanemann and Anthony C. Fisher
University of Calif., Berkeley

Break
Open Discussion Period

Workshop Wrap-Up
James J. Opaluch, University of Rhode Island

SESSION I. Economic Valuation in a Policy Context

COMPENSATION FOR NATURAL RESOURCE DAMAGES: AN EMERGING FEDERAL FRAMEWORK

by Roger C. Dower and Paul F. Scodari*

INTRODUCTION

Until quite recently, the most visible application of economic techniques for natural resources valuation was in the context of helping to guide policy decisions. For example, economic tools for measuring natural resource values have been used to determine the benefits of specific regulatory actions and to establish the impact of alternative development decisions. The passage of the Comprehensive Environmental Response, Compensation and Recovery Act in 1980 placed the spotlight on compensation for damages to natural resources as an alternative medium for the application of these tools. While there is a history under Federal and state statutory and common law of using economic analysis to value natural resource damages (especially from oil spills), CERCLA formalized this process by establishing a Federal regulatory and legal structure that will set the rules for how economic analysis will be factored into judicial proceedings involving oil and hazardous waste spills and releases.

The CERCLA compensation framework is just emerging and the regulations implementing the statute are being promulgated over the course of the next several months. The content and requirements of these regulations will have important implications for the allocation of social resources and perhaps for the field of natural resource economics. Further, the potential tensions between the theory of natural resource valuation and the practical constraints of an adversarial judicial system are bound to affect the ability of any compensation scheme to be equitable and cost-effective.

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The purpose of this paper is to provide an overview of the CERCLA natural resource damage assessment and compensation framework and to highlight several broad economic and legal issues that may affect the ability of the framework to achieve its intended purpose. By briefly reviewing CERCLA and its natural resource damage provisions as a whole; outlining the current assessment process; and identifying certain potential concerns, we hope to help provide a better understanding of the compensation process and foster a broader evaluation of the emerging framework. We should note one important caveat. The current damage assessment regulations have only been proposed. Final regulations are due in the near future and undoubtedly these will modify the current provisions, perhaps making some of our observations obsolete. Further, even if the regulations remain unchanged (which seems unlikely), many of the provisions will be the subject of judicial review as cases are brought under the statute. Court decisions on various aspects of the regulations could change their meaning and interpretation.

THE LEGAL STRUCTURE

Statutory Framework

1. Overview of the Act

In response to public conern over releases of hazardous substances into the environment heightened by the discovery of contamination at Love Canal, Congress enacted the Comprehensive Environmental Response, Compensation and Liability Act [1/] (CERCLA) in 1980 to deal with the threats posed by abandoned hazardous wastes sites and hazardous substances releases in general. The Act provides Federal and state governments with broad authorities to respond to past as well as future releases (actual or threatened) of hazardous substances into the environment. CERCLA also provides a liability and compensation mechanism for recovery of governmental response costs from the parties responsible for hazardous substance releases. To assure that money would be available to complete the job of cleaning up abandoned hazardous waste sites, CERCLA

established a $1.6 billion Hazardous Substance Response Fund financed primarily by excise taxes levied on crude oil and certain chemicals.

The basic liability and compensation provisions for response costs are set out in Section 107 of the Act. Liability is imposed on current and former owners and operators of polluting vessels or facilities, as well as those engaged in the generation, treatment, and disposal of hazardous substances [2/] for damages resulting from releases [3/] into the environment. The courts have interpreted these provisions as imposing strict, joint and several liability on these parties for hazardous substance releases. [4/] Essentially, this liability scheme can be used to force a "responsible" party to bear the full cost of cleaning up a hazardous waste release no matter how tenuous their connection to the release (or how many other parties contributed to the release) or how carefully they handled the offending wastes.

The $1.6 billion Hazardous Substance Response Fund [5/] was established to finance clean-ups in cases where the polluting parties are not known or are unwilling or unable to provide recompense. The types of claims permissable against the Fund include claims for payment of governmental response costs incurred under the Act's response authority provisions and other necessary response costs under the National Contingency Plan. [6/] Payment of claims by the Fund transfers to the Fund the right of the claimant to sue the polluting parties.

An important but often overlooked component of CERCLA is the Act's natural resource damage provisions. [7/] While the problem of cleaning up abandoned hazardous wastes sites has garnered considerable publicity, and a vast amount of litigation involving liability for response costs has occupied the courts, the potential significance of the natural resource damage provisions has generally escaped attention. However, these provisions have been called the Superfund "sleeper" and have the potential for greatly increasing the amount of damages polluting parties may be held liable for under CERCLA.

2. Natural Resource Damage Provisions

The CERCLA natural resource damage provisions authorize Federal and state governments to recover damages from polluting parties for the value of publicly owned natural resources injured by dicharges of oil or releases of hazardous substances. These provisions reflect Congressional recognition that hazardous substance contamination of the environment may impose social costs which would not be fully redressed by the clean-up of waste sites or private causes of action brought under state common law. The CERCLA legislative history suggests Congress' intent to allow for compensatory natural resource damages following existing common law doctrines. Together, the response cost and natural resource damage compensation provisions of CERCLA create a mechanism to force responsible parties to bear the full public costs and provide redress for their polluting activities.

Compensable natural resource damages are defined under Section 107 of CERCLA as damages for " . . . injury to, destruction of, or loss of natural resources, including the reasonable costs of assessing such injury, destruction, or loss resulting from such a release." 8/ The Act specifies that in the case of such natural resource injury " . . . liability shall be to the U.S. government and to any State for natural resources within the State or belonging to, managed by, controlled by, or appertaining to such State" and that "the president, or authorized representative of any State, shall act on behalf of the public as trustee of such natural resources, to recover for such damages." 9/ Natural resources are defined very broadly to include land, fish, wildlife, biota, air, water, ground water, drinking water supplies, and other such resources. 10/ The Act thus enables Federal agencies and state governments, who act as custodians for the public through protection and care of a wide range of publicly owned resources, to recover damages for injury to such resources caused by releases of oil or hazardous substances. CERCLA further specifies that "sums recoverable shall be available for use to restore, rehabilitate, or acquire the equivalent of such resources by the appropriate agencies of

the Federal government or the state government, but the measure of such damages shall not be limited by the sums which can be used to restore or replace such resources". 11/

The Act contains certain restrictions on compensation for natural resource injury. Section 107(f) provides that there can be no recovery for natural resource damages where the release of hazardous substances or oil causing the injuries occurred wholly before December 11, 1980. One court has held that defendents can escape liability for natural resource damages only if all release ended before, and no damages were suffered after, December 11, 1980. 12/ The Act further limits the liabilities of responsible parties for natural resource damages to $50 million. 13/

Trustees are also authorized to make claims against the Hazardous Substance Reponse Fund for natural resource damages in cases involving resource injury caused by hazardous substances. 14/ These types of claims can be made only for the costs of restoring or replacing the injured resource and only if the trustee bringing the claim has developed a restoration plan. 15/ While the Fund is theoretically available for such claims, it is likely that response actions will subsume most of the Fund resources and that trustees will be forced to sue responsible parties directly.

To assist trustees in bringing natural resource damage actions, Section 301(c) of CERCLA requires the President to promulgate regulations for use in guiding the assessment of natural resource damages. These regulations are to include two different types of standardized procedures for assessing natural resource injury and placing a dollar amount this damage: Type A or simplified assessment techniques for smaller releases; and Type B protocols that will include more detailed and extensive assessment methodologies for more major releases. Type A procedures are defined by the Act as "standard procedures for simplified assessments requiring minimal field observation, including establishing measures of damages based on units of discharge or release units or units of affected area". Type B procedures are specified by the Act to include "alternative protocols for conducting assessments in individual cases to determine the

type and extent of short- and long-term injury, destruction, or loss". The Act specified that these regulatons "shall identify the best available procedures to determine such damages, including both direct and indirect injury, destruction or loss and shall take into consideration factors including, but not limited to, replacement value, use value, and ability of the ecosystem or resource to recover". 16/ CERCLA also provides that damage assessments developed using these regulations will create a rebuttable presumption of accuracy. 17/

In summary, the CERCLA natural resource damage provisions create a powerful mechanism for the recovery of public damages resulting from natural resource injury caused by discharges of oil or releases of hazardous substances. By providing for damages from injury to a broad range of natural resources caused by many types of contaminants, and by providing a set of regulations to guide damage assessments bolstered by a rebuttable presumption of accuracy; the CERCLA natural resource damage provisions go beyond the scope of previous common law doctrines and statutes. 18/

Regulatory Famework

1. Implementation

The President, in Executive Order No. 12316, delegated to the U.S. Department of the Interior (DOI) responsibility for promulgating the Type A and Type B natural resource damage assessment regulations. This order also assigned to the U.S. Environmental Protection Agency (EPA) the task of promulgating procedures for trustees to follow for making claims against the Fund for natural resource restoration costs. However, the December 11, 1982 deadline imposed by the Act for promulgation of the assessment regulations and the claims procedures passed without either being published. In order to force the rulemaking process, the State of Montana filed suit against DOI and EPA for failure to perform their respective duties. This suit was subsequently voluntarily

withdrawn, but was followed by two new suits, one brought by the State of New Jersey and the other by the New Mexico Department of Health and the Environment.

The two suits were consolidated into one and heard in the District Court of New Jersey. 19/ As a result of a finding in favor of the plaintiffs on December 12, 1984, DOI entered into a consent order whereby the agency agreed to a specific timetable for the promulgation of the regulations. Under the consent order, DOI agreed to publish notices of proposed rulemakings for Type B regulations by December 20, 1985 and for Type A regulations by April 5, 1986. The consent order further specified that final Type B and A regulations would be published by April 22, 1986 and August 7, 1986, respectively. These deadlines have recently been extended slightly by the court. A notice of proposed rulemaking for Type B regulations was published in the Federal Register on the court imposed deadline. 20/ Proposed Type A regulations were published on May 6 of this year (51 Fed. Reg. 16636).

The proposed Type A regulations deal exclusively with damage assessments involving injury to coastal and marine environments. These assessment procedures make use of a computer model capable of mathematically calculating damages based on data concerning the types of discharges or releases of specific contaminants and the type of receiving coastal and marine resources. Because the model is only applicable to certain types of discharges and releases, and requires specific data on the resources affected, it can only applied to discharges into marine or coastal environments for which this type of data is available. At some future date, this system may be expanded or new systems developed to cover other types of resources and different types of discharges and releases. For now, however, damage assessments in cases involving injury to other types of resources are to guided by the Type B procedures.

The Type A regulations also provide trustees broad discretion in choosing between Type A and Type B procedures even in cases where the Type A procedures are applicable. Furthermore, potentially responsible polluting parties are given the option to

request Type B assessments even when Type A procedures are applicable provided they are willing to advance the costs to the trustee for performing the Type B assessment.

The remainder of this paper focuses only on the proposed Type B damage assessment regulations.

2. Overview of the Type B Regulations

The Type B regulations set out the basic processes to be followed by Federal and state trustees for: 1) determining and documenting natural resource injury caused by releases of oil or hazardous substances, 2) quantifying the effects of this injury on the human uses of the services provided by these resources, and 3) determining natural resource damages. The regulations explain the procedural steps for trustees to follow and provide criteria for selecting methodologies to determine resource injury and damages. They do not, however, provide specific guidance for implementing the various methodologies. This information is provided by a set of accompanying Technical Information Documents which evaluate and provide more specific information on the various alternatives. Because trustees are authorized to recover the costs of performing the assessment from the responsible parties, the regulations mandate that the assessment process be performed at reasonable cost. "Costs are reasonable when 1) the injury, quantification, and damage determination phases have a well-defined relationship to one another and are coordinated; and 2) the increment of extra benefits obtained by using a more costly injury, quanitification, or damage determination methodology are greater than the cost of that methodology." 21/

Our focus here is on the major provisions of the proposed regulations which contain key economic decision points or provide guidance for the use of economic methodologies for damage assessment. These provisions include Subpart B — Preassessment Phase, Subpart C — Assessment Plan, and Subpart E — Type B assessments.

a. Subpart B - Preassessment Plan - Subpart B of the proposed rule sets out the procedural steps for initiating the damages assessment process and for preliminary analysis of potentially injured natural resources. The most interesting section of this Subpart from an economic perspective is Section 11.23 which outlines the preassessment screen. The pre-assessment screen provides the criteria for determining whether the identified discharge or release justifies a natural resource damage assessment, and includes the first key economic decision point in the process.

The pre-assessment screen is defined as a "desk top" review of the existing data capable of being performed within a few days. The screen requires that a decision to proceed with an assessment should be based on the following determinations by the trustee: 1) that the discharge or release is covered under the relevant sections of CERCLA or The Clean Water Act 2) that the discharge or release has likely injured natural resources under the jurisdiction of the trustee; 3) that the quantity and concentrations of the contaminants released is sufficient to potentially cause resource injury; 4) that the data required to perform an assessment can be obtained at a reasonable cost; and 5) that any planned or completed response actions will not completely remedy the injury to the natural resources. The pre-assessment screen thus requires a preliminary determination by the trustee of the nature and extent of possible resource injury, based on the early sampling of contaminants and the area potentially exposed, as well as a determination of the human uses of the resources potentially affected. This information is to be used by the trustee to determine whether an assessment could be performed at reasonable costs, and the likelihood that a damages action would be successful.

b. Subpart C - Assessment Plan - After an affirmative decision is made to proceed in the pre-assessment screen, but before initiating a damage assessment, the trustee must develop a detailed assessment plan in accordance with the procedures set forth by Subpart C of the proposed regulations. Section 11.31 requires that the assessment plan identify all of the scientific and economic methodologies to be used in assessing the resource injury and determining damages in sufficient detail to be able to make a determination of whether the proposed assessment approach is cost-effective. The regulations interpret the term "cost-effective" to mean " . . . that when two or more activities provide the same level of benefits, the least cost activity providing that level of benefits will be selected." 22/

The Economic Methodology Determination section of the proposed rules (Section 11.35) allows the trustee to use restoration or replacement costs, or dimunition in use values as the basis for measuring natural resource damages and provides guidance to the trustee for the choice. This section specifies that "the authorized official shall select the lesser of 1) restoration or replacement costs or 2) dimunition of use values as the measure of damages." 23/ It further specifies that the costs and benefits of these alternative measures of damages be calculated based upon the readily available data and used to make this determination. The costs and benefits in this calculation are defined as "the expected present value, if possible, of anticipated restoration or replacement costs, expressed in constant dollars, and separated into capital, operating, and maintenance costs, including the timing of the costs"; and 2) "The expected present value, if possible, of anticipated use values gained through restoration or through replacement, expressed in constant dollars, specified for the same base year as the cost estimate, and seperated into recurring and nonrecurring benefits, including the timing of the benefit." 24/

The proposed regulation provide for an interesting exception to the above decision rule for "special resources". Special resources are defined as "natural resources that

have been set aside and committed to a specific use by law before the discharge of oil or release of a hazardous substance was detected. The term includes resources that were set aside primarily to preserve wildlife habitat or other unique and sensitive environments. It does not include resources listed on administratively determined lists for "special protection", or resources protected by regulatory status" or multiple use resources. 25/ In the case of special resources, the trustees may seek damages based upon restoration or replacement costs as long restoration or replacement of the resource is feasible and the costs "will not be grossly disproportionate to the benefits gained". 26/

The regulations further specify that the economic methodology determination should rely upon existing data and studies, and that no new data collection or modelling is needed to complete the determination. It also stipulates that if the existing data is insufficient to perform the economic methodology determination, it may be postponed until the completion of the formal injury determination phase in Subpart E. 27/

c. Subpart E - Type Assessments - Subpart E of the proposed regulations deals with the actual implementation of Type B assessments, and lays out the steps to be followed by trustees for choosing among and implementing alternative methodologies for each of the three major phases in the damages assessment process — injury determination, service reduction quantification, and damages estimation. The following discussion deals only with those parts of the Subpart dealing with the estimation of resource damages.

Guidance to trustees for estimating damages based upon restoration or replacement costs and certain restrictions on what these measures may include is discussed in Section 11.81. When restoration or replacement costs are to be used, they must be based on the least-cost alternative restoration or replacement scheme that returns the resource to its pre-injury baseline condition. Further, the restoration or replacement alternative used to calculate damages must be technically feasible to undertake. The measure of

damages calculated using restoration or replacement costs may also include any dimunition in resource use value over the recovery period. 28/

Criteria for the selection and implementation of use value methodologies is provided by Sections 11.83 and 11.84 of Subpart E. Key interpretations and definitions are also found here. The term "use value" is defined as "the value to the public of recreational or other public uses of the resource, as measured by changes in consumer surplus, any fees or other payments collectable by the government for a private party's use of the natural resource, and any economic rent accruing to a private party because the government does not charge a fee or price for use of the resource." Additionally, the regulations provide that, "In instances where the Federal or State agency acting as trustee is the majority operator or controller of a for- or not-for-profit enterprise, and the injury to the natural resource results in a loss to such an enterprise, that portion of the lost income from this enterprise . . . may be included as a measure of damages". 29/ Only the dimunition in value of baseline "committed uses" of natural resource services over the period it takes for the injured resource to actually recover can be used to measure damages. In addition, these baseline "committed uses" must be reasonable probable; purely speculative uses of the injured resource are precluded from consideration. 30/ A committed use of natural resource services is defined as "a current use, or a planned use of a natural resource for which the Federal or State agency acting as a trustee or another party has made a documented legal, administrative, budgetary, or a financial committment before the discharge of oil or release of a hazardous substance is detected." 31/

Section 11.83 of the regulations identifies and briefly describes the specific methodologies which may be used by trustees to estimate damages for both market and nonmarket natural resource services, and stipulates the conditions under which they may be used to estimate certain resource damages. An evaluation of these methodologies

which includes more specific information on their use is provided by an accompanying technical information document.[32/]

In the case of a resource for which a welldefined market exists, the regulations stipulate that the trustee must make a determination as to whether the specific market is reasonably competitive before choosing a valuation methodology. If the market for such a resource is reasonably competitive, the trustee should first turn to the market price methodology for detemining damages, which is based on the dimunition in market price for the injured resource. If the trustee determines that the market price methodology is not appropriate for valuing a particular resource, the regulations provide that the "appraisal" methodology be employed to value the resource if sufficient information exists. This methodology simply uses the difference between the before injury and after injury appraisal values for the resource in question. Trustees are instructed to turn to the "Uniform Appraisal Standards for Federal Land Acquisitions" for guidance in making such resource value appraisals. [33/]

The proposed regulations also provide for the use of specific methodologies for measuring the use value of nonmarket natural resource services. For nonmarket resources, the regulations allow the trustee to use various methodologies to estimate use value measures of damages based on estimates of either willingness-to-pay or willingness-to-accept (WTA). [34/] (Presumably, the regulations allow for WTA measures of damages because the property right for the resources covered under CERCLA are held by the public.) For injured resources which are used as inputs into the production of products associated with well-defined market prices, the factor income methodology may be employed to estimate the economic rent attributable to the resource as a measure of damages. Alternatively, for natural resources which provide consumer utility, the regulations specify the travel cost method, hedonic pricing, and the contingent valuation (CV) methods as acceptable approaches for measuring damages. [35/] While the regulations allow the trustee complete discretion in their choice of the travel cost or

hedonic pricing methods for measuring damages in the case of injury to nonmarket resources, certain restrictions are placed on the use of the CV approaches.

Section 11.83 (d)(5) explains that CV "can be used to survey consumptive, option, and existence value," but provides that "the use of this method to estimate option and existence values should be used only if the authorized official determines that no other valuation technique will be feasible." This stipulation can be interpreted as meaning that CV can only be used to measure resource values when no other valuation technique can be applied to estimate use values, and only under this condition can option and existence values (intrinsic values) be used as a basis for natural resource damages. This discussion is the only mention of intrinsic values in the regulations and suggests that DOI was unsure of how to handle these non-use values. Also, it suggests that DOI was uncomfortable with the CV methodology as a tool for measuring resource value.

Guidance is also provided on various smaller issues related to the implementation of the valuation methodologies. These issues include the handling of possible double counting problems; the treatment of uncertainty in damage determination; and discounting costs and benefits over time. The regulations specify that double counting of resource benefits should be avoided, but offer little guidance except to say that resource damages should be based on the residual resource injury after incorporating the effects, or anticipated effects, of response actions on resource services. 36/ With regard to the treatment of uncertainties in damage determination, the regulations state that when considerable uncertainties exist concerning the assumptions made when implementing valuation methodologies, trustees should consider alternative assumptions and document their effects on the calculation of costs and benefits. 37/ For discounting costs and benefits over time (including past and future), the regulations mandate the use of a 10% real rate of discount as specified by the Office of Management and Budget. 38/

SOME PRELIMINARY OBSERVATIONS

The natural resource damage assessment process should establish a framework to accomplish two important purposes. In an aggregate sense, the regulations should compensate the public for injuries to their natural resources from oil or hazardous waste spills and releases. Specifically, the scheme should make the public whole, so that the public is as well-off after natural resource injury as they were before the injury. The law should also seek to redistribute a specific subset of the costs of certain types of industrial or commercial activities: those non-health related external economic costs that fall on the public as a result of "improper" disposal or handling of oil and hazardous wastes. 39/ In this sense compensation, in combination with the other provisions of the Act, intends to fully internalize the social costs associated with past and future waste disposal practices. CERCLA assumes that the disposer is always in the position of being able to reduce risks most cheaply and thus bears the full responsibility of insuring against, and compensating for public natural resources injury.

In order for the CERCLA natural resource assessment process to achieve an efficient allocation of social resources, the system has to generate accurate estimates of the true economic value of injured natural resources and to do so while incurring the least costs possible. Given the nascent nature of state or Federal attempts to utilize the damage assessment scheme, it is difficult to forecast how closely it will hit the mark. Many elements of the assessment process will be modified and more fully defined by the court system as cases are heard and evaluated. However, there are several controversial elements, definitions and assumptions built into the damage assessment regulations that, assuming they are upheld by the courts, appear to undermine the equity and cost-effectiveness of the process. As currently written, the proposed regulations appear to suggest that natural resource injuries may be undervalued and the compensation process over-priced. These issues, as will be discussed below, are quite varied, but share one common characteristic. They all reflect, in part, tensions between the economic

concepts of natural resource value (and their estimation) and the constraints imposed by the legal environment in which these economic concepts will be brought to bear.

Public Versus Private Damages

The proposed regulations provide for the assessment of damages to publicly owned resources, but explicitly exclude compensation for injury to privately owned resources. The justification for this bifurcation of damage categories is the definition of resources covered by the Act which is interpreted by DOI to exclude damages that might be recoverable under private rights of action for injury to privately owned resources. Even in the absence of an actual natural resource damage case under these rules, this distinction between private and public resources has already generated tremendous confusion and controversy. Several key positions may help focus this debate if not provide ready resolutions. To help frame the following discussion, we refer to "private damages" as those private losses which result from injury to privately owned resources and define "public damages" as the aggregation of those private losses which result from injury to publicly owned resources.

o First, to an economist the distinction between public and private damages may seem somewhat arbitrary. If a hazardous waste release has altered the characteristic of a natural resource that serves as an input to the production of a recreational experience (utility function) or commercial product (production function), the economic damages are given by the willingness-to-pay (or sell) of recreationists or producers to avoid (accept) the additional cost of adjusting to the altered input. Whether the natural resource is privately or publicly owned is inconsequential at least on this level of analysis. However, accepting this distinction between public and private resources defined by the Act, economists would take the view that damages resulting from injury to publicly owned resources are represented by the aggregation of losses to all parties who use the resource.

In a strict legal setting, however, using the lost economic rents accruing to commercial harvestors of an injured animal species, for example, to place an economic value on that species, may suggest that private damages which are not permissable under the Act are at stake, regardless of the possible public trusteeship of the injured animals. The proposed regulations adopt, at least on the surface, the economic view of estimating public damages, which holds that private losses to individuals who use public resources represent the lost value of these resources. However, there appears to be an increasing tendency to interpret the rules to limit the use of private losses to individuals who use public resources to estimate for diminished public use values. To the extent that the rules change or that courts do not accept the economist's approach, it is possible that many uses of natural resource damage will go uncompensated under the rules.

o Second, is the issue of who is in the least-cost position for bringing successful natural resource damage claims. If the Act or the proposed rules limit the use by a public trustee of private losses as an approximation of public damage, private parties would be forced to bring individual suits under state common law to seek compensation. Given the subtlety of many of the injuries from hazardous waste spills and the expense and complexity of proper economic damage assessments, the conditions under which a private party could mount a successful case may be limited. A further complication results from damages to natural resources that do not obey property lines or political boundaries. It is not clear how injuries to private parties from contaminated air would be handled under a strict interpretation of private versus public resources. The risk of too narrowly defining public versus private damages is excessively large litigation and other transaction costs (such as duplicative assessments) to achieve fair compensation and possibly too few cases being brought. A more cost-effective solution might allow for consolidation of public and private damage claims when the trustee can take advantage of cost economies-of-scale.

o Finally, the economic notion of damages and their estimation does raise the issue of who gets the award. As the rules currently read, a public trustee claiming damages based on private losses has to use the award to restore, rehabilitate or acquire comparable natural resources. Private users would not receive compensation for losses incurred after the release but before the restoration even though such losses would be included in the assessment. From an economic point of view the resource allocation implications of such a distributional outcome are minimal. The legal questions are more interesting. For example, would a private party have standing to bring suit under state tort law against the state or the responsible party for damages incurred but for which they receive no compensation? The answer is unclear, but there may be some potential for double payments for the same injury to the extent that the private party is viewed by the courts as having a cause-of-action independent of the trustee's claim over the injured resource.

The debate over public versus private resources is not only a question of ownership. It involves distinctions between ownership, private versus public injuries, and private versus public losses. The net effect is uncertain pending resolution by the courts. In at least one case the court has ruled that private and public natural resources damages are best treated as one. However, the current bias would seem to be towards a more limited view suggesting higher transaction costs and fewer cases than may be warranted by the level of social costs involved.

The final assessment of this tension is an empirical one and has to await some practical experience with the process. It may be that the assessment process conducted by a public trustee will provide potential private parties with all the information and analysis they will need to bring compensation actions under state common law. On the other hand, the incentives for any one injured party to undertake the necessary studies to support a tort action on their behalf are sufficiently small to assume that few private actions will be brought in the absence of an organized group of plaintiffs or readily

identifiable and easily valued damages (such as fish kills). Is it in society's best interest to encourage a large number of relatively small but still expensive legal actions?

Intrinsic Values

The statute and the proposed assessment regulations allow for economic use values to be used as the basis for a natural resource damage claim. While the regulatory definitions appear to include both consumptive and non-consumptive uses of natural resources, the inclusion of intrinsic values such as option and existence values is not so clear. On its face, CERCLA could be read to include such losses when it defines the basis of a damage claim to include "but not be limited to" costs of restoration, lost use values, etc. The regulations also explicitly mention option and existence values but only in the context of using contingent valuation studies and then only when other techniques for measuring use value are not feasible. Other portions of the regulations limit economic damages to only those damages that can be associated with "committed" not "speculative" uses of the resource, perhaps ruling out the consideration of option and existence values in all but a few situations. On the other hand, there is a more subtle treatment of intrinsic values within the regulations when the economic decision rule concerning the choice of restoration costs or diminished use values is dropped for "special resources." The implicit recognition here is that there are some resources that would be undervalued if only consumptive and non-consumptive use values formed the basis for total economic value. The definition of special resource, however, is so narrow as to sharply limit its potential.

The apparent confusion in the regulations and growing debate among various interested parties concerning intrinsic values is the result of differences in perspective and philosophy. Some of these are more easily identified and discussed than others. Two examples may be helpful.

o First, is the issue of whether such values are true economic values. The answer here is somewhat a matter of philosophy; an environmentalist would say yes, an industrial polluter might say no. Yet, the evidence would seem to support a positive response. It is difficult to explain the high level of social resources that are devoted to protecting wilderness areas and endangered species; the creation of national parks and marine sanctuaries; and the whole of our nation's environmental protection efforts if option and existence values were not some part of the value we accord those programs. Further, the growing body of economic literature on the subject of intrinsic values, while not strictly in accord, does provide considerable weight at least on a conceptual level, as well as some empirical evidence for these values. 40/

o Second, is the question as to whether intrinsic values fall within the traditional legal concepts of economic value as defined under common law theories of damage. The case law involving natural resource damage cases is not very helpful here. We know of no single case where option and existence values formed an explicit basis for the damage claim. While some states have included such values in estimating damages to natural resource, we are not aware of any that have been the subject of court scrutiny, most are settled out of court and thus do not provide much in the way of precedent. 41/ However, a legal parallel may exist in personal injury cases. Courts have long held that in such cases compensation may be made for both direct economic losses (such as lost income, medical expenses, etc.) as well as "non-pecuniary" damages including pain and suffering, loss of consortium, and mental anguish over the loss of a loved one. 42/

Although not a perfect fit, intrinsic values have many of the same characteristics as the non-pecuniary damages in personal injury cases. Most notably, they both represent kinds of effects that we perceive to be real, but have a very hard time putting into dollar terms. There is an important difference between speculative damages (those that require a stretch of one's imagination to believe) and damages that are uncertain as to their value. While courts may be comfortable with the notion of pain and suffering as

a very real and believable effect of personal injury, they continue to grapple with how best to express those values in dollar terms. This uncertainty may not be the basis for excluding the consideration of "non-use" values, but may lead to widely varying outcomes for very similar cases.

o Finally, the uncertainty as to how best to value intrinsic damages leads to a third element of the controversy over their inclusion in natural resource damage assessments. While it may be theoretically possible to measure option and existence values through various economic methods, the technique with the most promise and that has been applied most often is contingent valuation. The question of whether one can accurately measure individuals' valuation of any commodity through preference revealing surveys continues to divide the economics community and is often dismissed out-of-hand by non-economists.

It is beyond the scope of this paper to venture into the contingent valuation debate except to note that there appears to be developing a consensus on the conditions that need to be met in order for a C.V. study to be credible, and an improved understanding of the limits and biases of the technique in its general application. 43/ Courts have extensive experience with judging the credibility of alternative approaches to measuring economic damages and a rather strong argument can be made that the option should continue to be available to public trustees (and others) subject to review and consideration of the judicial system.

There are undoubtedly other factors that carry weight in the intrinsic value debate. Once one is willing to admit that such values do exist under certain circumstances and that they are permissible under the law, the tensions between economics and the law would appear to collapse into a series of questions that courts have to deal with all the time to some degree. This would argue for more explicit recognition of the potential for such values in the regulations and the flexibility for the trustee to attempt to estimate such damages if they feel a credible case can be made.

The current trend in the proposed regulations and, it appears, at DOI is to limit the consideration of such values rather to expand it. Without offsetting changes in other parts of the regulations (such as broadening the definition of special resources) to provide some mechanism for incorporating such values on an indirect basis, there is a real possibility that many types of natural resources will go undervalued.

Real Versus Perceived Damages

The proposed regulations set out a process in which an injury to a natural resource is quantified and an economic value attached. The rules require that an injury be "measurable" in order for it to be part of a damage assessment. At the same time, the rules provide for the application of economic valuation tools in the damage assessment phase that do not necessarily require estimation of measurable injury (a trustee is still required to demonstrate the relationship between the pollutant and the damage). Because economists, value damages on the basis of changes in consumer and producer behavior, actual physical injury is assumed to result if one can isolate individuals' responses to a set of new conditions and resource characteristics. For an economist perceived damages are real damages if they result in changes to consumer utility or producer production capabilities.

It is not clear whether the authors of the proposed regulations were aware of this potential for conflict. Yet, given the bias of the legal system towards the demonstration of physical harm before the award of damages, the potential is very real. This is particularly so given the chronic, sub-acute nature of many of the environmental injuries that are likely to occur from hazardous waste releases. A wetland area containing above background levels of a particular pollutant (but below a state or federal standard) has experienced an economic damage if certain birdwatchers make fewer trips to the wetland. This is so even if there is no physical injury or risk of injury to the biological system. The proposed rules appear to accept this proposition, but at the same time

require a substantial demonstration of physical harm. Given what appear to be rather high burdens of proof concerning biological harm in the proposed regulations, it is not difficult to imagine a bias towards natural resource compensation only in those cases where clear evidence of physical effects is available.

CONCLUSION

The emerging compensation framework for natural resource damages attempts to integrate the economics and law of natural resource valuation into a single comprehensive package. The ability of the framework to achieve the dual goals of fair compensation at the lowest possible cost is a function of how carefully the perspectives and limits of economic valuation are coordinated with the constraints imposed by the legal structure in which these assessments will be judged. We have singled out three current issues concerning the application of natural resource economics to the CERCLA assessment process (as outlined in the recently proposed regulations) that appear to have the potential to skew the compensation formula towards under-compensation at relatively high costs. Others may well be equally important. For example, the reliance in the regulations on a 10 percent discount rate is a double edged sword, perhaps under-valuing future losses but over-valuing past damages.

We do not have a crystal ball that offers a clear picture of how the assessment process will actually be implemented in practice. Many of the issues raised here as well as many technical elements of the DOI proposed regulations await resolution in the final rules or in the courts. Nevertheless, we have tried to provide some hints as to likely outcomes. The magnitude of the possible effects cannot be predicted, but a qualitative assessment can be constructed. Our sense is that the tendency will be towards awards that represent something less than the full economic value of natural resources and involve relatively high transaction costs. Current litigation costs of CERCLA response cost recovery actions have been estimated to be as high as 40 percent of actual clean-up

costs (which EPA estimates to be around $7 million per site). The natural resource damage assessment process appears to risk raising these costs further.

The potential for the CERCLA assessment process to skew the costs and size of awards for natural resource damages places a high premium on incentives for the trustee to make decisions within the assessment framework that will lead to the most accurate estimates of value at the lowest costs possible. However, the basic decision rule imbedded in the proposed regulations is unlikely to provide much in the way of guidance to the trustees. The proposed rules require the trustee to seek a balance between the costs and benefits of the assessment, but benefits are defined in terms of the size of the award. Everything being equal, the larger the award for a given cost, the more justified is the assessment. A more appropriate decision rule for choosing methodologies would define benefits in terms of the value of additional or new more accurate estimates of damages and the reallocation of resources resulting from that award. Of course, it is possible that the current decision rule will foster large awards and thus offset some of the more negative impacts discussed above. Yet, the net result is hard to predict and may bias the outcome of the rules even more. While we acknowledge the tremendous difficulty in assigning dollar benefits to information, there would seem to be rather high payoffs from investigating alternative specifications for the current decision rule.

FOOTNOTES

1/ 42 U.S.C. §§9601-9657 (hereinafter cited as "CERCLA").

2/ CERCLA §101(14) defines "hazardous substance" primarily by referring to designations made under a variety of other environmental statutes. These hazardous substances include 696 substances (see 48 FR 23552) as well as any additional substances designated by EPA pursuant to §102 of CERCLA.

3/ CERCLA §101(22). "Release" is broadly defined but excludes exposures resulting from specified nuclear materials, workplace emissions, most engine exhausts, fertilizer applications and "federally permitted releases." (See §101(10))

4/ Reed, P.D., "CERCLA Litigation Update: The Emerging Law of Generator Liability," 14 ELR 10024 (1984).

5/ CERCLA §221.

6/ CERCLA §111(a)(11)-(4).

7/ The CERCLA natural resource damage provisions are contained in three separate sections of the Act: §107, §111 and §301.

8/ CERCLA §107(a)(4)(C).

9/ CERCLA §107(f).

10/ CERCLA §101(16).

11/ CERCLA §107(f).

12/ 546 F. Supp. 1100, 12 ELR 20954 (D. Minn. 1982).

13/ CERCLA §107(c)(1)(C).

14/ CERCLA 111(b).

15/ CERCLA §111(i).

16/ CERCLA §301(c)(2).

17/ CERCLA §111(h)(2).

18/ The provisions of various Federal Laws relating to natural resource damage compensation are reviewed in: Yang, E., R. C. Dower, and M. Menefee, The Use of Economic Analysis in Valuing Natural Resource Damages, U.S.Department of Commerce, Washington, DC 1984.

19/ State of New Jersey et al. v. Ruckelshaus et al., Cir. No. 84-1668 (D.C.N.J.)

20/ 50 Fed. Reg. 52127 (Dec. 20, 1985) (hereinafter cited as "Regulations").

21/ Regulations §11.14(ee).

22/ Regulations §11.14(j).

23/ Regulations §11.35(a)(2).

24/ Regulations §11.35(e)(3)(i) and (ii).

25/ Regulations §11.14(pp).

26/ Regulations §11.35(d)(2)(i)-(ii).

27/ Regulations §11.35(e)(1) and (2).

28/ Regulations §11.81(a)-(f).

29/ Regulations §11.83(b)(1) and (2).

30/ Regulations §11.84(b)(2).

31/ Regulations §11.14(h).

32/ Desvouges, William H., Type B Technical Information Document: Techniques to Measure Damages to Natural Resources, Draft Report Prepared for the CERCLA 301 Project, U.S. Department of Interior, September 1985.

33/ Regulations §11.83(c)(1) and (2).

34/ Regulations §11.83(d)(7).

35/ Regulations §11.83(d)(1)-(5).

36/ Regulations §11.84(c).

37/ Regulations §11.84(d).

38/ Regulations 11.84(e).

39/ For an economic analysis of compensation see, Viscusi, W. K., "Alternative Approaches to Valuing Health Impacts of Accidents: Liability Law and Prospective Valuations," Law and Contemporary Problems, Vol. 46 (1983), pp. 49-68.

40/ See, for example: Desvouges, W. H., J. K. Smith and M.P. McGivney, A comparison of Alternative Approaches for Estimating Recreation and Related Benefits of Water Quality Improvements, U.S. EPA; Wash., D.C., 1983; and Walsh, R. G., J. B. Loomis and R. S. Gillman, "Valuing Option, Existence and Bequest Demands for Wilderness," Land Economics, Vol. 60 (May, 1981).

41/ The historical treatment of natural resource damages in courts and a review of past cases can be found in Yang, E., R. C. Dower and M. Menafee, The Use of Economic Analysis in Valuing Natural Resource Damages, U.S. Department of Commerce: Wash., D.C., 1984.

42/ A general treatment of damage categories for tort actions is found in Prosser, W. C. and J. W. Wade, Torts 2nd - Restatement of the Law, American Law Institute: St. Paul, 1979.

43/ A current overview of contingent valuation methods is found in, Cummings, R.O., D.S. Brookshire and W. Schulze, "Valuing Public Goods: The Contingent Valuation Method, Totowa: Rowman & Allanheld, 1986.

Measuring Damages to Coastal and Marine Natural Resources from Oil and Hazardous Substance Spills: Application of An Integrated Ocean Systems/Economic Model

Thomas A. Grigalunas, James J. Opaluch*, Deborah French and Mark Reed**

Introduction

The Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA), and the Clean Water Act (CWA) as amended, establish polluter liability for the costs of responding to and cleaning up spills of oil or hazardous substances covered by these Acts and for the costs of assessing damages to natural resources. In addition, the Federal government and the States, in their roles as trustees, can claim damages for injuries to natural resources. CERCLA requires the Federal government to promulgate two types of regulations for assessing damages to natural resources: type A regulations are to provide standard procedures for simplified assessments requiring minimal field observations, and type B regulations which specify alternative protocols for conducting assessments in individual, site-specific cases (Sec. 301(c)(2)). Hence, the Act recognizes that damage assessment studies can be quite costly; the simplified, type A assessment is intended to apply to cases for which an incident-specific, type B estimate of natural resource damages is judged not to be worth the cost.

In addition to its distributive implications (i.e., compensating governments as trustees for natural resources), the liability provisions of CERCLA can have important resource allocation effects. Liability for damages is akin to a Pigouvian tax on externalities, and recent research suggests that liability can provide incentives for controlling stochastic pollution events (e.g., Opaluch and Grigalunas, 1984). As recognized in

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the Clean Water Act, liability has the potential "... to create incentives to achieve a higher standard of care in all aspects of the management of hazardous substances ...". Liability is one of the few examples where federal environmental policy uses financial incentives, which economists typically argue are potentially more cost effective then traditional Command-and-Control regulations. However, assessing damages from an incident can be extremely costly and may bring to question the cost effectiveness of liability rules. For example, estimating the social costs from the AMOCO CADIZ oil spill cost approximately $6.6 million. Clearly, this magnitude of expenditures can be justified only in the relatively rare case of a catastrophic incident.

Further, much of the injury which occurs may not be readily observable, particularly for marine spills, where many dead organisms sink, disperse, or are rapidly eaten by scavengers. For example, approximately $1.4 million ($1986) was spent to evaluate the consequences of the 179 thousand barrel ARGO MERCHANT spill, but no injury was found. For most relatively modest marine incidents damages may not be observable; and hence it may not be desirable to base liability on observed damages.

To be effective, the assessment process must be relatively quick and inexpensive to administer, and must not be based only on damages which are readily observable. This paper discusses an alternative approach for measuring liability for damages from pollution incidents based on the concept of a damage function. A model which runs on the IBM PC (or compatible) is constructed which simulates the dispersion of a pollutant through the environment and the resultant injury to biological communities. The model then provides an economic measure of damages from this presumed injury without the need to carry out a damage assessment involving expensive field observations. This framework is currently being considered by the U.S. Department of Interior for use in measuring damages to coastal and marine environments for the relatively small incidents which would call for a type A natural resource damage assessment.

Following a brief description, the model is applied to measure the damages from hypothetical oil and hazardous substance spills in selected coastal and marine environments. Because the draft study described in this paper is in the review process, which may lead to refinements of the

model and the data, the analysis and results presented must be regarded as preliminary.

Due to space constraints, the discussion in this paper will be extremely brief. The interested reader is refered to the draft technical report upon which this paper is based (Economic Analysis, Inc. and Applied Science Associates, 1986).

II. Overview of Methodology and Data

Clearly, the consequences of a given oil or hazardous substance spill could vary greatly, depending upon the amount and characteristics of the substance spilled, such as its physical and toxicological properties, and the characteristics of the environment in which the spill occurs, such as the location and season of the incident, the water depth, currents, temperature, and the specific natural resources in the affected area. The measurement of damages from a particular incident requires that linkages be established, in sequence, from an incident covered by CERCLA or the CWA, to its effect on ambient conditions, to biological and physical injuries and, ultimately, to the measure of damages which is quantified in monetary terms. An integrated, interdisciplinary model provides an operational framework for quantifying these linkages. As depicted in Figure 1, the model is comprised of three submodels: the physical fates, biological effects, and economic damages submodels.

The physical fates submodel has a chemical data base which contains information on several hundred chemicals obtained from established data bases. The physical and toxicological data contained in this data base includes such parameters as density, solubility, vapor pressure, degradation rates in sea water and in sediments, octanol/water partition coefficient (Kow), adsorbed/dissolved partition coefficient (Koc), and toxicological information for phytoplankton, zooplankton, ichthyoplankton, adult fish, and benthos.

Given the amount and the physical/chemical parameters of the substance spilled, the fates submodel simulates its spreading, mixing, and degradation in four layers of the environment: the surface, upper water column, lower water column and bottom. In addition, the submodel accounts for the amount of pollutant lost to the atmosphere through evaporation, where appropriate. A mass balance calculation ensures that

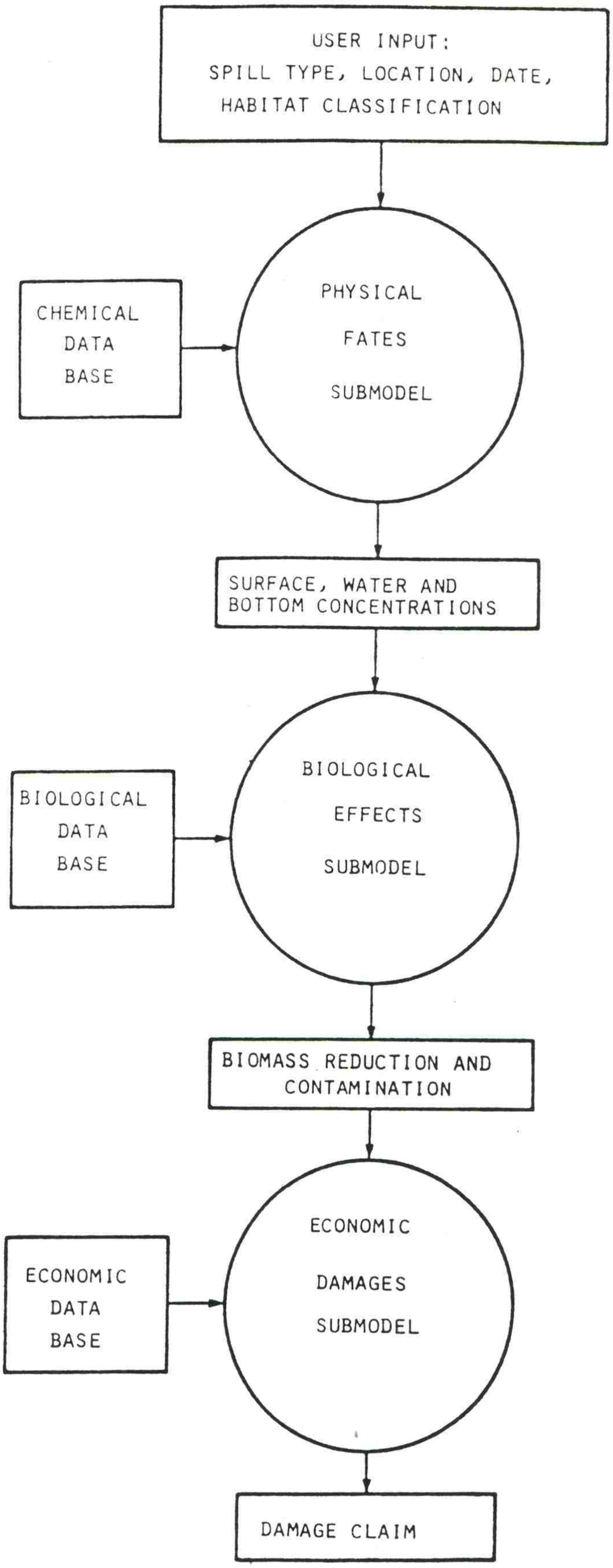


Figure I Overview of type A natural resources damage assessment model for coastal and marine environments.

the sum of the mass of the pollutant in all environmental compartments at each point in time equals the mass spilled.

To simulate the fate of a spilled oil or hazardous substance, the physical fates submodel incorporates information on specific coastal and marine environmental parameters. These parameters include the mean and tidal currents, wind speed and direction, depth of the upper water column, depth of the lower water column, as well as the air and water temperatures and distance to shorelines, or boundaries of concern, in each direction. In a particular application, these environmental parameters can be set by the user; otherwise the model employs default values for each parameter.

The output of the physical fates simulation is concentration of the pollutant, over time, in various cells for each of the four layers. This information is passed to the biological submodel, which calculates injury to various biota in the environment. To define biological resources in contact with the spill, the biological submodel employs a substantial data base on biological abundance of various categories of finfish, shellfish, marine mammals (fur seals), and birds. The data base specifies the abundance of species groups in each of 10 provinces/ecosystem types defined in Cowardin et al. (1979) for the marine environment of the U.S. and its territories (Figure 2). Abundance of the species groups vary by season, bottom type, marine vs estuarine, and tidal vs subtidal environments. In total, 91 different ecosystem catagories are considered in the biological submodel.

The effect of a spill on marine organisms depends on the concentration of the substance in the physical environment where the organisms live. Above a threshold level, the impact increases with concentration, using the results of standard laboratory toxicity test data. The biological submodel calculates direct loss of adult and juveniles for waterfowl and shorebirds and fur seals and for nine fish and shellfish species catagories and loss of larvae for each of these catagories. In addition, a simple trophic model is used to trace indirect losses through the food chain.

Biological injury quantified in the submodel includes (1) short-term injury (e.g., death) and (2) long-term injuries which occur over time (e.g.,reduced recruitment). Three categories of short-term biological effects are considered. First, surface slicks (e.g., oil) may be

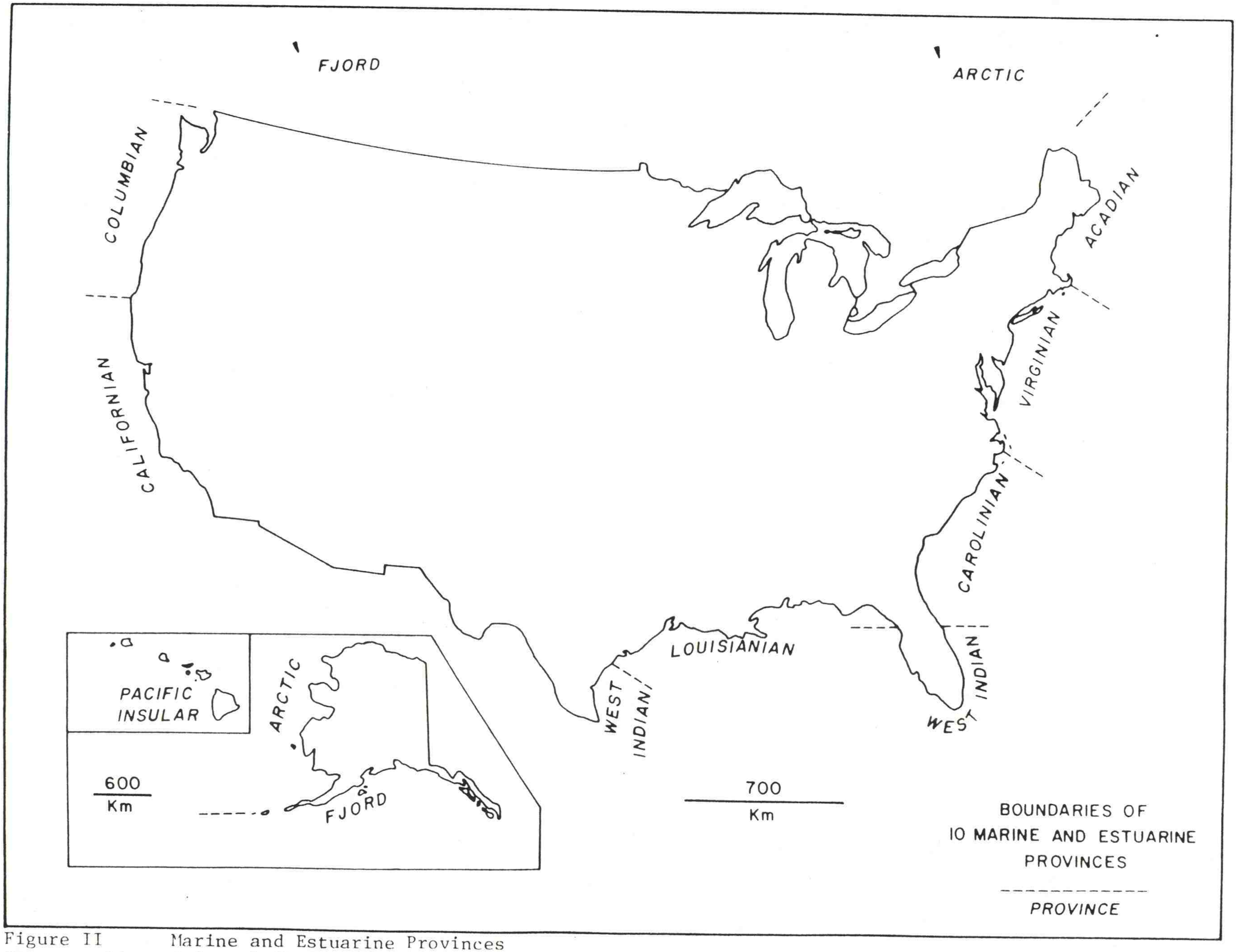


Figure II Marine and Estuarine Provinces
Source: Adapted from L.M. Cowardin et al., 1979. "Classification of Wetlands and Deepwater Habitats of the United States," FWS/OBS-79,31.

encountered by birds and fur seals. Second, the dissolved portion of a spill can kill various fish species. Finally, spilled material can sink to the bottom, killing bottom fish species.

Long-term losses due to the effects of acute toxicity on the productivity of the biomass also are taken into account. The dynamics of the biological system is traced using the Ricker model (Ricker, 1975) which simulates the dynamics of cohorts, or age classes, of organisms by calculating changes in biomass due to changes in numbers of individuals within a cohort through natural and fishing mortality, as well as the change in biomass due to growth of individuals within the cohort. The dynamics of the number and weight of indiviuals within a particular age class are described as:

$$N(t) = N(0) \exp[-(M+F)\ t]$$

$$W(t) = W(0) \exp[G\ t]$$

respectively, where N(t) is the number of individuals within an age class at time t, M is the natural mortality rate, F is the fishing mortality rate, W(t) is the weight of a representative individual in the age class at time t, and G is the natural growth rate. Thus, the biomass of a particular age class at time t is:

$$B(t) = N(t) * W(t) = N(0)*W(0)*\exp[(G-M-F)\ t]$$

and the total biomass of a fishery is:

$$X(t) = \sum_{i=t_0}^{T} B(i)$$

where $X(\cdot)$ represents the total biomass of all age classes in the fishery, t_0 is the age of recruitment to the fishery and T is the maximimum life span of the fish.

Using a simple bioeconomic model, lost catch from a spill can simulated over time as:

$$H^{NS}(t)-H^{S}(t)= q^{NS}E^{NS}X^{NS}(t)-q^{S}E^{S}X^{S}(t) = F^{NS}X^{NS}(t)-F^{S}X^{S}(t) \quad (1)$$

where H is total catch, superscript NS represents the case with no spill and S represents the case with the spill, q is the catchability coefficient and E represents the level of fishing effort applied. The resultant lost total discounted economic rent to commercial fisheries, for example, is:

$$\pi^{NS}-\pi^{S} = \int_{t_s}^{\infty} \left[\{p^{NS} H^{NS}(t) - c^{NS}E^{NS} - [p^{S} H^{S}(t) - c^{S}E^{S}]\} \exp(-rt) \right] dt$$

where t_s represents the time of the spill, p is the ex-vessel price of the fish, c represents cost per unit effort, and r represents the discount rate. Thus, in general, all variables must be allowed to change as a result of the pollution incident. However, since the methodology is meant to be used for relatively small spills some simplifying assumptions are possible. First, small spills are unlikely to cause changes in market prices of fish, the catchability coefficient or in cost per unit effort. Hence, these are assumed to be constant with and without the spill. In addition, small spills are unlikely to have a substantial impact on the level of effort applied to the fishery as a whole. In addition, very little work has been done on the issue of effort response (e.g. Bockstael and Opaluch (1983)) and predictions of changes in effort would be difficult, at best, without an incident- specific study to consider alternatives available for the particular individuals impacted by the spill of concern. For these reasons, fishing effort is presumed to be unaffected by the spills of concern.

Using the methodology described above, long-term commercial and recreational fishery losses due to the effects of acute toxicity on the biomass are considered. The output of the biological submodel is a time series of lost catch for species groups for fin and shellfish, as well as losses in various groups of birds and fur seals.

Indirect biological losses quantified in the submodel fall into two categories. First, larvae and juveniles may be killed, resulting in long term losses through eventual reduction in recruitment. Second, spills may

kill lower food chain organisms which have no commercial or recreational value but which contribute to predator species which do have economic value. A food chain model specifically allows for an assessment of incident-specific damages via predator-prey relationships.

Once the short-term and long-term biological injuries have been quantified following a particular incident, damages can be measured. The measure of damages is defined as the present value of the lost in situ use value of the injured natural resources over the time period through resource recovery. The categories of coastal and marine natural resource damages considered, and the general relationship of the economic damages submodel to the other submodels are illustrated in Fig. 3.

Damages resulting from injury to lower trophic, non-commercial organisms are based on the ultimate loss in the in situ use value of predator species (commercial and recreational fisheries, waterfowl and shorebirds, and fur seals) which occurs when an incident affects the productivity of the food chain. The food chain or ecological model developed in the biological effects component of the ocean systems model is incident-specific and quantifies the biological injuries to predator species which arise over time as a result of the incident. Given the quantification of biological injuries, damages are measured using the concepts and data applicable to commercial and recreational fisheries, to waterfowl and shorebirds, and to fur seals, outlined below.

In order to measure lost in situ use value, fish resources injured by an incident must be allocated between commercial and recreational harvests foregone. Lost in situ value for commercial fisheries is the change in the total value of landings minus the change in the cost of harvesting the fish (i.e., lost economic rent). For recreational fisheries, lost in situ value is the loss in sportsfishing benefits due to the reduced catch rate from smaller stocks, net of the change in the cost of catching the fish (i.e., lost consumer surplus).

A standard bioeconomics model is used to derive the measure of damages to commercial and recreational fisheries. The short-term and long-term and direct and indirect injury to fish which would have been harvested in the absence of the incident is an output of the biological effects submodel. Injured species are allocated between commercial and recreational uses, given estimates of the relative weight of recreational and commercial landings, by species, for each province and of the total

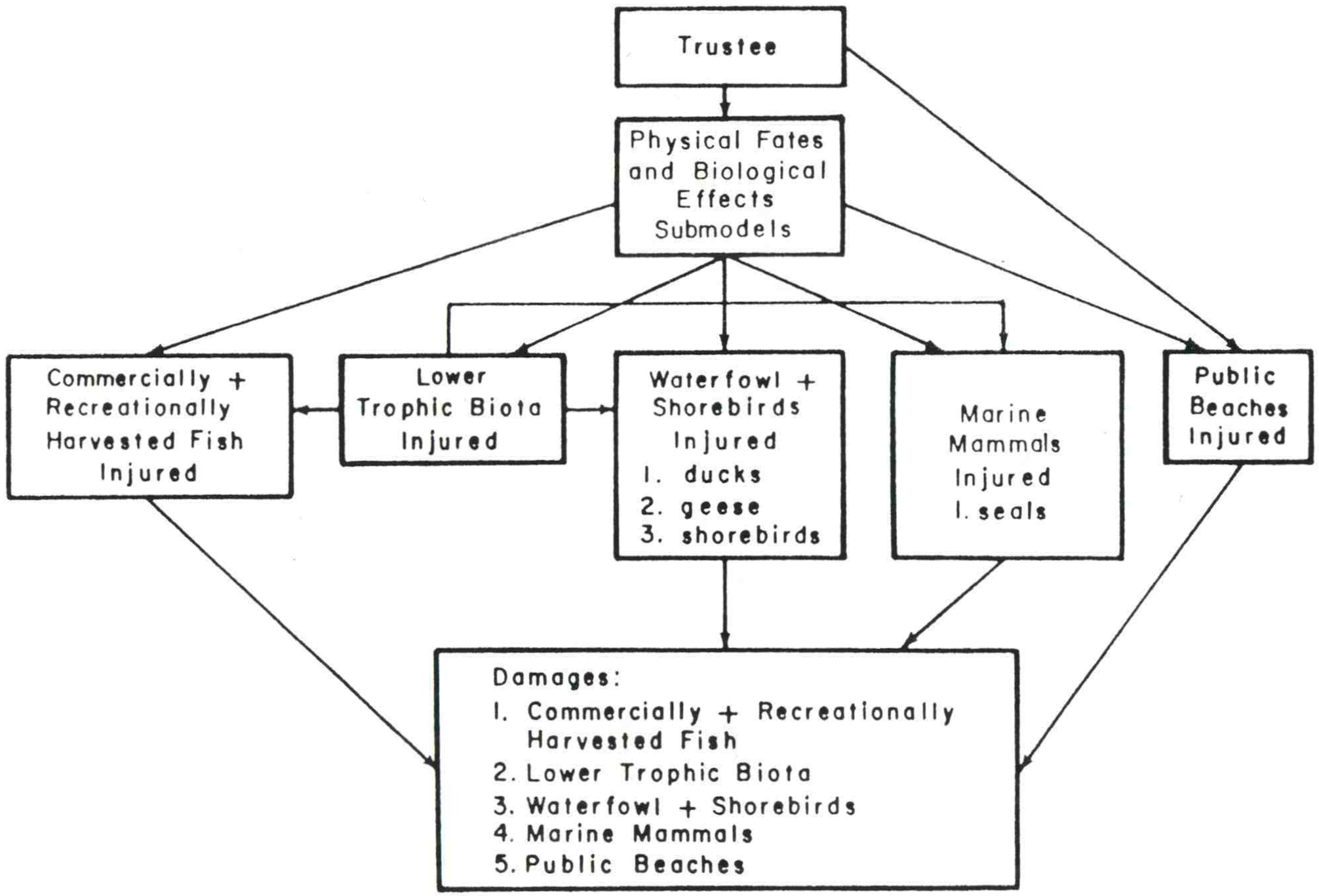


Figure XII Simplified representation of the natural resource damage assessment process and damage categories considered in economics component for the coastal and marine environment for CERCLA Type A incidents.

fishing mortality rate for species groups. Total fishing mortality is broken into commercial and recreational fishing mortality as:

$$F_{TOT} = F_{COM} + F_{REC} = q_{COM} E_{COM} + q_{REC} E_{REC}$$

Since

$$H_{COM} = q_{COM} E_{COM} X = F_{COM} X$$

and

$$H_{REC} = q_{REC} E_{REC} X = F_{REC} X$$

Commercial and recreational fishing mortality rates can be calculated as:

$$F_{COM} = H_{COM}/(H_{COM} + H_{REC}) F_{TOT}$$

$$F_{REC} = H_{REC}/(H_{REC} + H_{COM}) F_{TOT}$$

Hence given estimates of total fishing mortality rates and commercial and recreational catch, lost stock can be allocated among lost recreational and commercial catch over time using Equation (1) above.

Ex-vessel (price at the dock) fish prices, averaged over 1982-1984, are used to evaluate damages to commercially harvested fish. Province-specific price information for commercial fisheries and catch data for commercial and recreational fisheries are from National Marine Fisheries Service sources. Values for recreational sports fishing are adapted from the literature.

Injury to waterfowl and shorebirds results in losses of consumptive (hunting) and non-consumptive (e.g., viewing, photographing) in situ use values. The quantification of biological injury to waterfowl and shorebirds is an output of the biological effects model. Damages resulting from consumptive use value losses are measured using available estimates of the marginal value of an additional waterfowl (duck or geese) harvest. Using the results of Brown and Hammack (1977) damages arising from non-consumptive use value losses for non-game species are measured by employing an estimate of the marginal change in visitor days

associated with a change in bird population for a wildlife refuge. The resulting estimate of lost visitor days then are evaluated based on a unit day value published by the Water Resources Council (1979).

IV. Discussion of Selected Preliminary Results

This section describes applications of the model to a variety of environments and substances spilled. The esturarine spills presented in this section assume that all incidents take place at a location where the water depth is 30 feet; the pycnocline (separating the upper and lower water columns) is assumed to be at 15 feet. The marine spills presented all assume water depth of approximately 120 feet with the pycnocline assumed to be 60 feet. Except where otherwise indicated, all spills are assumed to take place on mud bottoms during the summer season when the surface water and air temperature are assumed to be 25 and 24 degrees Centigrade, respectively. Again it is important to note that because the draft study described in this paper is under review, which may lead to refinements of the model and data, the results contained in this section must be regarded as preliminary.

The sample runs were chosen to provide a perspective on how the results change as the major characteristics of hypothetical incidents vary. Since oil is by far the most common substance spilled, the base case incident is the 100 metric ton (750 bbl) oil spill. Sensitivity analyses were run in which the quantity of oil spilled varied from 40 metric tons (300 bbl) to 1,000 metric tons (7,500 bbl) in both marine and estuarine environments to indicate how damages vary with the quantity spilled and in marine vs. estuarine environments. Additional cases were run to examine how damages vary with province in which the spill occurs and the season of the spill. For this purpose, the base case 100 metric ton estuarine spill was run in all ten provinces and in one province (the Virginian province) during each of the four seasons. Also, two cases of onshore (intertidal) spills were run to indicate damages from spills which come ashore.

But spills don't have to occur in the intertidal to eventually reach there, do they?

Additionally, a series of runs is provided to examine releases of non-oil substances with different physical properties to indicate the sensitivity of damages with respect to the characteristics of the

substances. The oil cases discussed above provide a perspective on floating substances. Hence, additional cases are run on a sinking substance and on a substance which mixes readily in the water column. Each of these cases is run for spills of 50 and 100 metric tons.

Table 1 contains the model output for the base case 100 metric ton oil spill. As can be seen, the largest categories of damages to finfish occur to demersal fish (e.g. flounder) at $58.7 thousand, piscevorous fish at (e.g. striped bass, bluefish) at $54.2 thousand and semi-demersal fish (e.g. cod) at $49.6 thousand. The damages to mollusks (e.g. oysters, clams, scallops) total $42.4 thousand and damages to decapods (e.g. crabs) total $24.4 thousand. Also, substantial damages occur to waterfowl ($2.5 thousand). Using a discount rate of 10 percent as required by Office and Management and Budget Circular No. A-94 (as revised), the present value of the total damages from this 100 metric ton spill are $241.6 thousand.

The results for the damages as a function of the quantity of oil spilled in July in the Virginian province are presented in Table 2 and in Figure 4. As can be seen, oil spill damages increase with quantity spilled, and do so at an increasing rate. However, damages become more closely linear as quantity spilled increases. A spill of 300 barrels of oil in an estuary leads to losses of $34 thousand. Increasing the quantity spilled to 1500 barrels leads to damages of nearly $850 thousand. Hence, for this 5-fold increase in amount spilled in an estuary, damages increase by a factor in excess of 25. Increasing quantity spilled to 7,500 barrels results in damages of nearly $8 million.

Thus, damages from oil spills exhibit rapid increase with the quantity spilled, such that the damage function is clearly nonlinear. This suggests that a fixed charge per barrel spilled would be inappropriate for oil spills, as both the average and the incremental damages per barrel increase as the quantity spilled increases, particularly for relatively small spills.

The second set of sensitivity analyses examines damages from the base 100 metric ton oil spill in each of the ten provinces. As can be seen in Table 3, the highest level of damages occurs in the Louisian province

Table 1

Lost In Situ Value of Commercial Plus Recreational Fin Fish in the Virginian Province from a 100.0000 Metric Ton Spill of Prudhoe Bay Crude Oil (HEAVY - 20% volatiles)

YEAR	Anadromous Fish	Planktivorous Fish	Piscevorous Fish	Top Carnivores	Demersal Fish	Semi-Demersal Fish
1986	18.33	866.32	3243.65	.00	739.78	9075.97
1987	17.77	795.38	2998.12	.00	679.37	8251.66
1988	14.41	1207.82	4156.92	.00	617.60	7501.51
1989	11.59	999.84	3697.79	.00	7748.87	6535.24
1990	9.23	851.73	3324.97	.00	7269.06	4379.76
1991	7.26	744.28	3022.83	.00	6868.05	3052.22
1992	5.62	664.55	2778.57	.00	6518.15	2212.83
1993	4.27	603.78	2581.67	.00	6203.07	1664.76
1994	3.15	556.09	2423.50	.00	5913.12	1293.41
1995	2.23	517.49	2297.00	.00	5642.50	1031.65
1996	1.48	485.28	2196.36	.00	5387.60	839.81
1997	.88	457.65	2116.83	.00	5146.16	694.05
1998	.39	433.35	2054.51	.00	.00	579.88
1999	.00	411.53	2006.22	.00	.00	488.24
2000	.00	.00	1969.33	.00	.00	413.27
2001	.00	.00	1941.74	.00	.00	351.10
2002	.00	.00	1921.67	.00	.00	299.03
2003	.00	.00	1907.72	.00	.00	255.10
2004	.00	.00	1898.70	.00	.00	217.88
2005	.00	.00	1893.69	.00	.00	186.23
2006	.00	.00	1891.87	.00	.00	159.26
2007	.00	.00	1892.67	.00	.00	136.25
2008	.00	.00	.00	.00	.00	.00
Total	96.61	9595.09	54216.33	.00	58733.33	49619.11

Total Discounted Commercial Plus Recreational Loss for All Species of Finfish (Rounded to the Nearest Dollar) ...$ 172260.

Table 1 (Continued)

Lost In Situ Value of Commercial Invertibrates in the Virginian Province from a 100.0000 Metric Ton Spill of Prudhoe Bay Crude Oil (HEAVY - 20% volatiles)

YEAR	Mollusks	Decapods	Squid
1986	2189.18	3841.46	1.03
1987	2023.80	3492.24	1.51
1988	1839.82	3174.76	.98
1989	1672.56	2886.15	.48
1990	3140.53	2623.77	.00
1991	2835.69	2385.24	.00
1992	2586.84	2168.40	.00
1993	2383.03	1389.84	.00
1994	2215.46	890.53	.00
1995	2077.04	570.63	.00
1996	1962.07	365.68	.00
1997	1865.97	234.37	.00
1998	1785.03	150.23	.00
1999	1716.30	96.32	.00
2000	1657.40	61.77	.00
2001	1606.41	39.63	.00
2002	1561.78	25.44	.00
2003	1522.29	16.34	.00
2004	1486.94	10.50	.00
2005	1454.95	6.76	.00
2006	1425.67	4.36	.00
2007	1398.59	2.82	.00
2008	.00	1.83	.00
2009	.00	1.19	.00
2010	.00	.78	.00
2011	.00	.51	.00
2012	.00	.34	.00
2013	.00	.23	.00
2014	.00	.15	.00
2015	.00	.11	.00
2016	.00	.07	.00
2017	.00	.05	.00
2018	.00	.04	.00
2019	.00	.03	.00
2020	.00	.02	.00
2021	.00	.02	.00
2022	.00	.01	.00
2023	.00	.01	.00
2024	.00	.01	.00
2025	.00	.01	.00
2026	.00	.01	.00
2027	.00	.01	.00
2028	.00	.00	.00
2029	.00	.00	.00
2030	.00	.00	.00
2031	.00	.00	.00
2032	.00	.00	.00
2033	.00	.00	.00
2034	.00	.00	.00
2035	.00	.00	.00
2036	.00	.00	.00
2037	.00	.00	.00
2038	.00	.00	.00
Totals	42407.35	24442.66	4.00

Total Discounted Loss for All Species of Invertibrates
(Rounded to the Nearest Dollar) ...$ 66854.

Table 1 (Continued)

Lost Value of Birds and Marine Mammals in the Virginian Province from a 100.0000 Metric Ton Spill of Prudhoe Bay Crude Oil (HEAVY - 20% volatiles)

Year	Seals	Waterfowl
1986	.00	2477.77
1987	.00	.00
Totals	.00	2477.77

Total Discounted Loss for Seals and Birds (Rounded to the Nearest Dollar) ...$ 2478.

Total Discounted Losses For All Categories		
Fishery Losses	- $	239114.
Bird and Fur Seal Losses	- $	2478.
Total for All Categories	- $	241592.

Table 2 Damages From Oil Spills As A Function of Quantity Spilled

Quantity		Total Damages ($000)	Damages per Barrel	Incremental Damages per Barrel Increment
Metric Tons	Barrels			
40	300	$34	$113.80	$0.11
60	450	$84	$185.68	$0.33
100	750	$242	$322.12	$0.53
200	1500	$848	$565.01	$0.81
400	3000	$2,380	$793.31	$1.02
800	6000	$5,998	$999.71	$1.21
1000	7500	$7,998	$1,066.36	$1.33

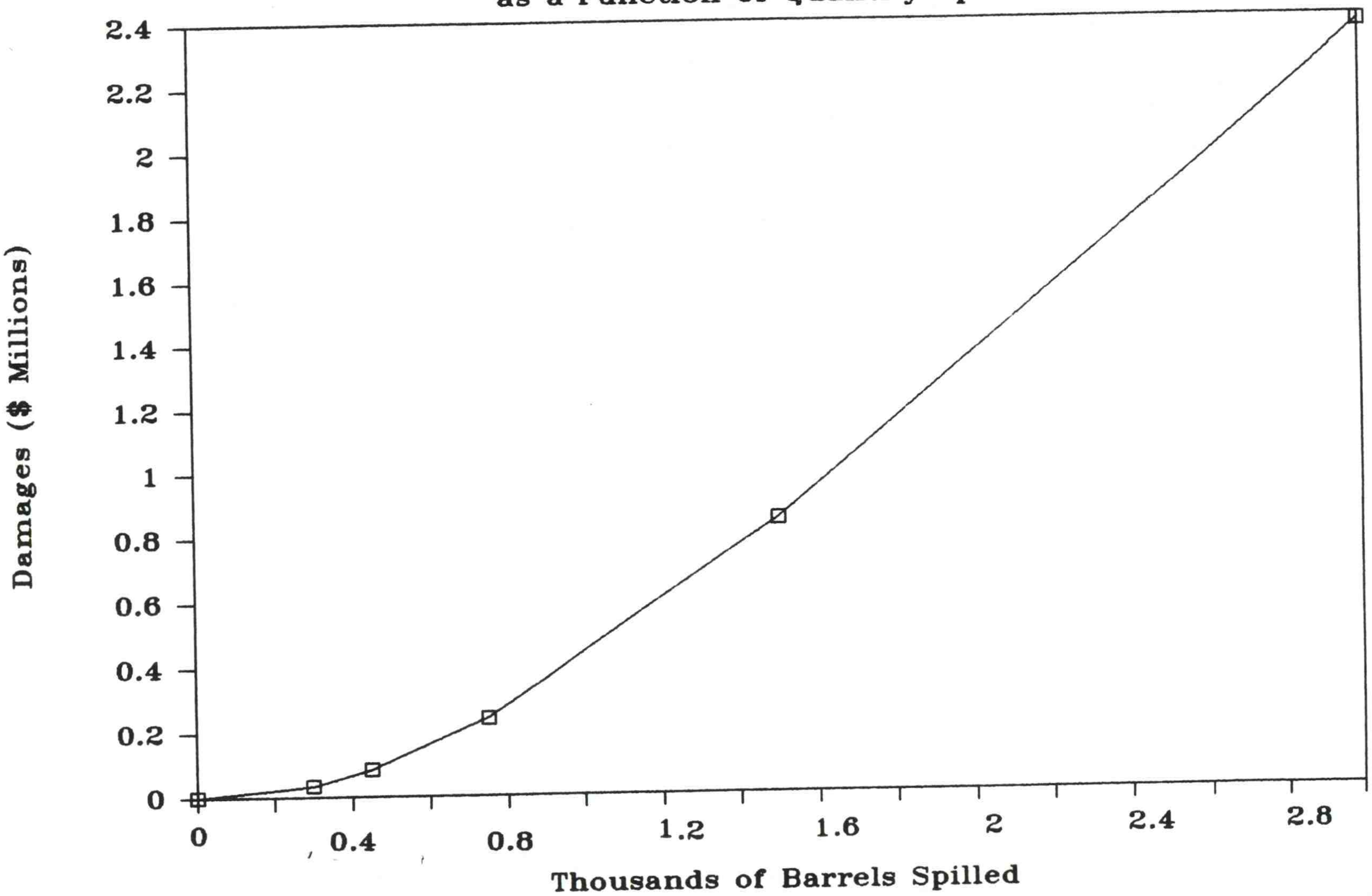
Table 3 Damages from a 100 Metric Ton Oil Spill in Estuaries in Each Province

Province	Damages ($000)
Louisianan	$ 1,508
Californian	$ 921
Columbian	$ 411
Carolinian	$ 847
Fjord	$ 307
Arctic	$ 282
Acadian	$ 242
Virginian	$ 242
Pacific Insular	$ 160
West Indies	$ 75

Figure 4

Damages from Oil Spills

as a Function of Quantity Spilled



($1,507 thousand) followed by the Californian ($921 thousand), the Carolinian ($847 thousand), the Columbian ($411), the Fjord ($307 thousand), the Arctic ($282 thousand), Acadian ($242 thousand), the Virginian ($242 thousand), the Pacific Insular ($160 thousand) and the West Indies ($74 thousand).

The next set of sensitivity analyses examines damages over seasons. Table 4 contains the aggregate results of the model for spills in an estuarine environment in each of the four seasons. As can be seen, damages are highest in the summer ($241 thousand), followed by spring ($123 thousand), fall ($86 thousand) and winter ($41 thousand).

Table 5 contains the results for the intertidal spills. The first intertidal spill occurs on a sandy shore and results in damages of $42 to fisheries, $192 to shorebirds, and $40.5 thousand to public beaches. Beach damages were presumed to result in closure of 1 kilometer of beach for one week. The second intertidal case was presumed to occur on a rocky shoreline. For this case, fishery damages total $80 thousand and damages to shorebirds total $1,745.

The final set of sensitivity analyses examines damages from substances which sink and which mix into the water column, with each substance considered for a 50 and 100 metric ton spill. Again, all spills in this analysis are assumed to occur in a Virginian, estuarine environment during the summer season. The characteristics of the substances and results for these cases are presented and compared with crude oil in Table 6.

The sinking substance is pentachlorophenol (PCP), which has a density of 1.98, is relatively insoluble and has a half life of about 90 days in seawater. PCP is quite toxic with an LC50 of 115 ppb for fish. Total damages for a 50 metric ton spill of PCP is $57 thousand and for a 100 metric ton spill damages total approximately $127 thousand. Damages increase with the amount of PCP discharged; but in contrast with the case for crude oil, damages do not appear to be highly non-linear. This occurs because for sinking substances the material released falls to the bottom rapidly, thus the area impacted will increase only slightly with the amount of substance released. In contrast, oil will spread out rapidly, covering a substantially larger area as the amount spilled increases.

Sulfuric acid represents a substance which mixes readily with seawater. It is more dense than seawater (1.84), is highly water soluble

Table 4 Damages from a 100 Metric Ton (750 Barrel) Oil Spill Across Seasons

Season	Damages
Spring	$123,280
Summer	$241,592
Fall	$85,692
Winter	$41,021

Table 5 Total Damages to All Categories from a 75 Metric Ton Spill of Prudoe Bay Crude Oil (20% Volitiles) in an Estuarine, Intertidal Environment in the Virginian Province

	Sandy Shoreline	Rocky Shoreline
Fishery Losses -	$ 42.	$ 80035.
Bird and Fur Seal Losses -	$ 192.	$ 1754.
Damages to Public Beaches -	$ 40481.	$ 0.
Total for All Categories -	$ 40715.	$ 81789.

Table 6 Comparison of Properties and Damages from Spills of Various Substances in an Estuary in the Virginian Province.

Property	Prudo Bay Crude Oil	Pentaclorophenol	Sulfuric Acid
Density	0.89	1.98	1.84
Solubility	1.00E+00	8.00E+01	1.00E+06
Degradation Rate	1.00E-03	7.70E-03	1.90E-03
Octonol/Water (Kow)	1.00E+01	3.10E+03	1.00E-03
Adsorbed/Dissolved (Koc)	1.00E+03	3.71E+02	1.00E-01
LC50 for Fish	1.00E+03	1.15E+02	7.50E+04
Damages			
@ 50 Metric Tons	$57,079	$57,355	$72
@100 Metric Tons	$241,592	$127,261	$174

and has a relatively low level of toxicity (LC50 of 7.5 ppm for fish). Due to a high solubility, sulfuric acid is rapidly diluted in the water column, and quickly becomes non-lethal to marine biota. Damages for the 50 metric ton discharge total $72 and for the 50 metric ton discharge damages total $174.

V. Summary and Concluding Comments

In summary, various properties of substances released have dramatic effects on the level of damages caused by the release, as well as the shape of the relationship between damages and the quantity of the substance released. Further, for a given substance, damages are sensitive to the location and the season of the discharge or release. This suggests that an approach which sets liability as a function of characteristics of the substances must consider how the substance behaves in the environment, and not simply descriptors such as the level of toxicity and the amount spilled. This was made clear by comparing the case of oil discharges, which rapidly spread, with that of sinking substances released, where the area impacted is more restricted. For the latter case, releases in the range of 50-100 metric tons resulted in relatively constant average damages per ton of PCP as the amount released increases. In contrast, average damages per barrel of oil discharged increase with increasing amounts spilled. An approach which assigns liability to spills without regard to considerations such as the location, season or transport of the pollutant can easily be in error concerning the relationship between damages and quantity released, or other characteristics of the pollution incident.

VI. References

Bockstael, N.E. and J.J. Opaluch. 1983. "Discrete Modelling of Supply Response Under Uncertainty: The Case of the Fishery" Journal of Env. Economics and Management (June) 10(2).

Brown, Gardner M. and J. Hammack. "Commenwealth v. Steuart, Economic Valuation of Waterfowl" Unpublished manuscript (mimeo) May 5, 1977.

Cowardin, L.M., V. Carter, F.C. Golet and E.T. LaRoe. 1979. Classification of Wetlands and Deepwater Habitats of the United States. Fish and Wildlife Service, U.S. Dept. of Interior.

Economic Analysis and Applied Science Associates. 1986. Measuring

Damages to Coastal and Marine Natural Resources: Concepts and Data Relevant for CERCLA Type A Damage Assessments Draft document prepared for CERCLA 301 Project, U.S. Department of Interior.

Opaluch, J.J. and T.A. Grigalunas. 1984. "Controlling Stochastic Pollution Events Through Liability Rules: Some Evidence from OCS Leasing." Rand Journal of Economics (Spring) 15(1).

Ricker, W.E. 1975. "Computation and Interpretation of Biological Statistics of Populations" Bull. Fish. Res. Board of Canada, 191(382).

U.S. Congress. 1980. Comprehensive Environmental Response, Compensation and Liability Act of 1980. Public Law 96-510, 96 Cong. 94 (Dec. 11, 1980) Stat. 2767-2811.

U.S. Congress. 1982. The Clean Water Act (as amended through December, 1981). Public Law 97-117, 97th Cong. (February, 1982). Washington, D.C., U.S. Government Printing Office.

Water Resources Council. 1979. "Unit Day Value Method."Federal Register, Vol. 44 No. 242.(Dec. 14).

Economic and Environmental Conflicts in OCS Oil and Gas Leasing: An Analysis of National vs. Regional Benefits and Costs

by

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INTRODUCTION

A now commonplace chapter in Principles of Economics textbooks points out that smoke from a factory which damages nearby crops or noise from airplanes which overfly residential areas in the process of landing constitute economic externalities--costs borne not by the firms causing the injuries nor by their customers, but by third parties; costs not accounted for in the usual market process.

Looking more deeply into this issue, a further point might be made that an increasingly important category of external costs does not involve actual damages to third-parties but, instead, merely the risk of damages. Risk-bearing of this type lowers real income for affected individuals and communities just as surely as conventional damage; we know this to be true from many studies of consumer behavior.

But a particularly troublesome aspect of risk-bearing (and one seldom acknowledged in the literature) is the fact that the costs involved are often not determined by the actual risk of harm to the individual but merely by the perception of risk. The significance of this principle in relation to environmental policy is apparent. If the public comes to believe that the risks of a government program affecting the environment are high (whatever the true risks), the expected political outcome will be that implementation of the program will be made more difficult or stopped entirely. Indeed, one of the most effective methods of opposition to programs sponsored or regulated by the government, including nuclear power, has been to convince the public that the risks associated with the program are unacceptably high while at the same time ignoring the offsetting reduction of other (and often greater) risks which the program would provide.

This method of opposing government programs is particularly effective if it is supported by the political leaders within an affected region. As an example of this principle, consider the case of the recent efforts by the Department of Interior to accelerate the pace of leasing of lands on the Outer Continental Shelf for oil and gas development. Opponents of an expanded leasing program have managed to stifle the Interior Department's efforts by exploiting the well-known principle that whenever a

conflict exists between national and local benefits from a public program, most politicians will put local interests ahead of the national interest.

The degree to which public perceptions of risk may be altered by misinterpretation of scientific evidence is exemplified by a recent experience in California where U.S. Geological Survey researchers reported findings in 1982 suggesting that the Mammoth Lakes area had a high probability of experiencing a major earthquake. Although the report did not say this, the media misinterpreted the findings as predicting an earthquake in the immediate future. This sent the local real estate market into a tailspin, reducing property values by about 40 percent over the next three years. While the actual risk of earthquakes was unaffected by the USGS report, the perception of risk was dramatically heightened. And once this perception took hold, it was almost impossible to dispel it, despite vigorous attempts by the USGS researchers to downplay the practical significance of their research after they recognized the impact it was having.

Similarly, in the first fourteen years of leasing federal lands offshore for oil and gas development (that is, prior to the Santa Barbara oil spill of January 1969), OCS lease sales inspired little comment and no serious protests. In the years since the Santa Barbara spill, however, the public response to proposed OCS lease sales has been almost uniformly negative. This misperception by the public of the true risk of oil spills has heightened in recent years, largely because of legal and political opposition mounted by state and local governments (typically with the enthusiastic support of the local media) in California, Massachusetts, Alaska, Florida, and other coastal states. As a result, the Interior Department's plan for greatly escalating the pace of OCS leasing has been effectively blocked and some 52 million acres of potentially productive OCS lands have been declared off limits for leasing by means of legislatively imposed moratoria.

One of the remarkable aspects of this situation is the fact that state and local objections to federal leasing have no reference point in party politics. Leasing opponents include both conservative Republicans like Governor George Deukmejian of California and Senator Paula Hawkins of Florida, and their more liberal political opponents, Mayor Tom Bradley of Los Angeles and Governor Bob Graham of Florida. Indeed, the public opposition to OCS leasing has become so nearly universal in some states that it seems no longer to require the support of scientific evidence but has taken on the character of a quasi-religious crusade.

The oil and gas resources of the Outer Continental Shelf represent a major economic asset to the nation whose utilization can contribute to economic growth, national security, and the

improvement of real income for all Americans. If the true benefits and costs of enhanced development of OCS oil and gas resources are not being correctly balanced in current deliberations over leasing policy, what needs to be done to change this situation so that the full benefits of these resources can be secured for the nation? This is the essential issue that this paper will attempt to deal with.

I. THE ECONOMIC BENEFITS TO SOCIETY OF OCS LEASING

Economic benefits flow to American society as a result of OCS oil and gas leasing in three major ways:

1. Economic Rent. For all OCS lands leased in total, the market value of expected oil and gas resources exceeds expected costs of production. The difference is referred to by economists as economic rent. Competition among bidders for OCS leases results in the transfer of this economic rent to the American people as owners of these resources.[1] As indicated in Table 1, lessees of OCS lands had paid the federal government $59 billion in bonus, royalty, and other payments for OCS resources through December 31, 1982 (or 56 percent of the total cumulative production value of these resources). In 1982 dollars, the total amount of rent collected amounts to $90.4 billion. About half of the royalty payments included in this figure (or about 15 percent of the total) were derived from production of natural gas. But it is impractical to look for or produce natural gas from the OCS independently; thus OCS oil and gas may essentially be considered to be joint products. Thus in balancing the risk of environmental damage in relation to collection of economic rent, it is appropriate to include all economic rent collected from OCS oil and gas on the benefit side. Looked at this way, it can be seen that the federal government, through 1982, collected $15.00 in economic rent (1982 dollars) for each barrel of oil produced on the OCS.

2. Marginal Import Premium on Petroleum. It is often assumed in theoretical discussions that the elasticity of supply of petroleum imports into the U.S. is infinite. In truth, given the share of world imports accounted for by the U.S., any increase in U.S. demand for imported oil has some impact on the price paid,

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[1]Economic rent is maximized when competition in the market for leases induces winning bidders to pay all but the necessary costs of production, including a normal profit, over to the lessor (the federal government). Studies by Mead, et al., have shown that winning bidders for OCS leases have earned no more than normal rates of return on their OCS lease investments, suggesting that the federal government has received full economic value for the leases issued. See [7], Table 3-1, p. 53.

particularly in the short run. A study of this impact made by the Energy Vulnerability Modeling Project at Stanford University estimated that the "monopsony buying power wedge" for the U.S. was $7.70 per barrel in 1980 [8, p. 13]. A more recent analysis suggests that the marginal import premium (or the excess of import costs per barrel above actual price) is about $6.00 per barrel under plausible assumptions about recent market conditions [12, p. F-34]. The $6.00 per barrel figure would appear to be a conservative measure of the import premium associated with additional production of crude oil from the OCS since the premium is being attributed only to oil production when, in fact, both crude oil and natural gas produced from the OCS have the effect of offsetting petroleum imports.

3. Enhancement of National Security. Two aspects of the national security argument are considered below. The first is based strictly on considerations of national defense; the second considers the broader issue of economic security.

a. National Defense. Few would deny that petroleum supply is a critical requirement for a country in a time of war or national emergency. While the validity of the arguments supporting protection of the domestic oil industry on national defense grounds has often been questioned, Congress has at various times endorsed policies designed to promote the security of domestic petroleum supply using various legislative approaches such as tax subsidies for domestic oil producers, the Mandatory Oil Import Quota System, and the Strategic Petroleum Reserve.

b. Economic Security. An interruption in the supply of imported oil has the potential to severely disrupt the functioning of the U.S. economy, as the experience of the period following the Arab oil embargo of 1973 indicates. Estimates of the potential economic disruption premium in social cost of imported oil range up to $100 per barrel [2, p. 54], but these estimates are highly speculative and, in any case, include both real and pecuniary external costs. A more conservative approach is to assume that the benefits to U.S. economic security gained as a result of enhanced OCS oil and gas production are about the same, on a per barrel basis, as those which are obtained from the existing Strategic Petroleum Reserve. The full economic security benefits of the SPR are not known, but we may assume they are at least as great as the SPR's current opportunity cost, which is estimated to be between $3.25 and $4.00 per barrel [3, p. 235]. We will accept the latter figure as as a low-range estimate of the national security benefits of enhanced OCS oil production.

The listing of benefits given above is probably conservative because it ignores possible benefits of enhanced productivity of U.S. labor and capital in OCS development and production and makes no allowance for the impact of OCS development on the U.S. balance of payments, both of which would likely result in higher

real income for Americans. Furthermore, no account is taken of possible tecnological spillover benefits from OCS development which could be captured by U.S. exploration and drilling companies (benefiting the nation both directly and through enhancement of export income to U.S. oil service companies).[2]

In summary, the benefits of enhanced OCS oil production include collection of economic rent by the federal government, reduction in the price premium on imported oil, and national security benefits. Together, these benefit categories amount to about $25 per barrel, on average, expressed in 1982 prices.

It must be emphasized that these benefits are in addition to the value of the oil in the marketplace. They represent a true social surplus or premium above resource cost. From the point of view of national economic policy, they are the benefits which should be contrasted with the estimated external social costs of OCS development in determining whether enhanced OCS development is economically justified.

II. THE EXTERNAL SOCIAL COSTS OF OCS OIL AND GAS DEVELOPMENT

No environmental policy issue, with the exception of nuclear power, has received deeper or more comprehensive study in the U.S. than that of offshore leasing and development. Through 1983, the Department of Interior alone had spent over $340 million on environmental and related studies of offshore development, and announced plans to continue such studies at the rate of about $30 million per year in succeeding years [13, pp. 48, 82, and 121-23]. Additional studies of offshore oil and gas development and oil spill impacts have been carried out or funded by the the Environmental Protection Agency, the U.S. Department of Commerce (NOAA), and other federal and state agencies. At this point, there is little likelihood that critical information relating to the environmental costs of offshore oil development--information of such gravity that it might significantly alter the scientific outlook on the issue--has yet to be discovered.

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[2]Paul Kobrin has shown that if OCS resource development is merely delayed rather than foregone, a significant social cost is incurred because delay in receipt of economic rent diminishes its present discounted value (see [4]). The Department of Interior has used this approach in estimating the cost of OCS moratoria. It should be noted that this analysis of the cost of delay is valid only if the proposed rate of production is optimal in a capital theory sense (i.e., each unit of production over time has the same expected present value).

Nevertheless, concerns continue to be raised about the environmental and other impacts of OCS development, usually in these categories of potential social cost:

1. Impairment of water quality resulting from disposal of drilling muds or cuttings, or of water produced from wells.

2. Impairment of air quality resulting from loading, unloading, or processing of oil and gas.

3. Oil spill impacts on commercial fishing.

4. Oil spill impacts on tourism and recreation.

5. Oil spill impacts on the natural environment.

6. Oil spill impacts on property values.

7. Uncompensated costs of oil spill cleanup.

8. Infrastructure costs to local governments.

These areas of concern will be addressed individually in the sections which follow.

1. Drilling Muds, Cuttings, Produced Water. The possible impact of drilling muds and cuttings on the marine environment has been of major concern to both industry and government, leading to research expenditures of about $15 million in recent years. These studies conclude that environmental impacts of drilling muds are localized, temporary, and minor (see [13], pp. 86-87; [5], p. 725; [1], pp. 119-21; and [9], pp. 100-03). Only low toxicity drilling muds (all approved by the EPA) are used offshore. Some turbidity is created by offshore drilling, but it is of little significance in comparison to the impact of of conventional dredging activities or clam harvesting. Studies both in U.S. waters and the North Sea show that drilling muds have only minor ecological consequences within a limited area around the platform. Similar conclusions are reported in studies of "produced water", which under strict environmental stipulations, must be treated and separated before it can be reinjected or discharged from the platform.

2. Air Quality. Environmental stipulations attached to OCS leases require use of "best available control technologies" on production, transportation, and storage facilities to protect air quality. Both the EPA and state air pollution control agencies are involved in development of stipulations and in monitoring of compliance, and these agencies have full access to platforms for purposes of inspection or testing (see [13], pp. 84-85; and [1], p. 125).

3. Impacts on Commercial Fishing. Studies by the present author of the impact on commercial fisheries of three major oil spills (Santa Barbara, USA, 1969; Zoe Colocotroni, Puerto Rico, 1973; and Amoco Cadiz, France, 1978)--only one of which resulted from offshore drilling--concluded that in no case did the oil spill cause any major loss to open seas fisheries (see [6], [10], and [11]). Damage to shell fisheries in bays and estuaries was significant in the Amoco Cadiz case, but this damage was quite easily quantified and thus the damaged parties were more quickly and adequately compensated. Oil spills have their major impact on open seas fisheries by preempting access to the fishing grounds, but the studies noted above found that the catch lost during shut downs was more than fully recovered in later fishing efforts. The same conclusion was reached by European researchers studying impacts of oil spills on fisheries in the North Sea. Commenting on these studies, the Royal Commission on Environmental Pollution declared, "we agree with the findings...that the effect of oil pollution in general on adult fish populations is so slight as to be undetectable in fishery statistics" [9, p. 54].

4. Impacts on Recreation and Tourism. Potential impacts of oil spills on tourism are generally exaggerated by opponents of OCS leasing, who often suggest that oil operations offshore have the potential to "destroy" an invaluable onshore tourism industry. In fact, only limited damage to tourism from oil spills has been documented. Researchers in Britain discovered no impacts on tourism as a result of two major oil spills occurring near tourist destination sites on the coast of Cornwall (Torrey Canyon and Newquay). They concluded that "most people accept oil pollution as just another uncertainty relating to a seaside holiday," and that factors such as bad weather or other forms of coastal water pollution, particularly sewage, are "much more significant" than oil pollution in affecting tourism [see 9, p. 56]. Losses to tourism were also insignificant in the Zoe Colocotroni, Argo Merchant, and Ixtoc spills. Only two oil spills appear to have affected tourism in any major way: Santa Barbara and Amoco Cadiz. In both cases, some shifting of tourists from affected to unaffected sites nearby was detected (meaning that social costs to the local economy were greater than to the state or national economies). Also in both cases, unusually bad weather was experienced at the time of the oil spill, making it difficult to identify the impact of the oil spill alone. As might be expected in view of the unique qualities of Santa Barbara and Brittany as destination sites, impacts on tourism were temporary.

5. Impact on the Natural Environment. This is the most controversial aspect of oil spill damage, and the one which clearly elevated the Santa Barbara spill above all others in public significance. A considerable body of research has documented the damage to marine organisms caused by oil spills, particularly in the cases of Torrey Canyon, West Falmouth, Santa Barbara, Zoe Colocotroni, and Amoco Cadiz. Later studies have

shown, however, that the natural environment in each area has either completely recovered or is in the process of recovering. No long-term damage to the ecosystem has been documented. It is beyond question that the environmental damage observed in Santa Barbara, particularly in respect to bird losses, was shocking to the public. But is was, nevertheless, temporary. How should the social cost of this type of loss be evaluated? At this point, economists have not developed a completely convincing and practical methodology.[3] Most economists accept the idea that monetary values should be attached to the amenity or psychological losses suffered by society when living organisms are killed or injured as a result of oil spills.[4] A conflict in values arises, however, when economists try to sell this idea to environmentalists who may insist that living organisms should be accorded unique or paramount value over all other categories of social cost. This conflict cannot be resolved by debate, but it is worth noting that society routinely accepts various risks to human life in all areas of human activity and has accepted, as well, the need for (and appropriateness of) economic approaches to evaluating actual or potential loss of life. Is it not reasonable to accept a similar approach, using economic criteria, in evaluating the worth of other living things?

6. Impacts on Property Values. Our study of the economic cost of the Santa Barbara oil spill concluded that property values of private homes in the beachfront area had declined temporarily as a result of the spill, with a consequent loss of economic rent to owners [6, p. 225]. A similar investigation of the question of potential loss of property values in the area of Brittany most impacted by the Amoco Cadiz oil spill, however, showed that no discernible losses had occurred [11, p. 77]. This result is not surprising since few buyers of real estate in that area of Brittany are speculators and an oil spill, even in the light of recent history, is an exceptional and temporary event. Beachfront property owners in California are among the most vociferous opponents of OCS leasing, but the potential for damage to property values from offshore oil development cannot be taken too seriously in view of the risks routinely faced by such owners from earthquakes, mudslides, fires, and flooding. Realistically, the oil spill risk is of negligible significance.

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[3] A review of some of the methodologies used in evaluating environmental damage and their theoretical and practical difficulties is provided in [11], pp. 81-85.

[4] The response of the oil industry to the public outcry over environmental damage caused by the Santa Barbara oil spill has been dramatic. In the entire period since 1970, less than 800 barrels of oil have been spilled as a result of well blowouts on OCS leases (see [13], p. 3).

7. Uncompensated Costs of Cleanup. Fears that coastal residents or government agencies would be saddled with the costs of cleaning up oil spills left behind by unknown or insolvent parties led to a spate of legislation in the early 1970's, resulting in the establishment of oil spill compensation funds in several states and at the federal level. To this date, no losses from oil spills related to OCS development have been paid for out of these funds. This is an expected result, since spilled crude oil can be fingerprinted and OCS operators, even if they wanted to, could not steal away from the scene of a spill. The record of payments from the Florida Coastal Protection Fund over the most recent six-year period indicates that only $20,000 per year has been paid out for all cleanup and damage claims. Indeed, the Fund balance of about $40 million has been accumulating interest so rapidly in recent years that the Legislature has had the luxury of diverting part of it to handle inland and groundwater problems in Florida. The Offshore Oil Pollution Compensation Fund created by the OCS Lands Act amendments of 1978 has also been faced with only a small total value of claims, despite its capitalization level of $200 million. Some may not want to accept the fact, but phantom oil spillage is quite unusual and not very expensive to deal with. Significant oil spills are not easily covered up, and responsible parties are usually quick to act in payment of cleanup costs.

8. Infrastructure Costs. This category of damages is highly suspect since most local governments assiduously seek out new industries expecting to recover for any short-run costs through expansion of their tax base. If the proposed developments have particularly high start-up costs to local governments (as might be true for a major new OCS oil storage or transport facility), the best way of dealing with this situation is through use of impact fees or special assessments. Reviewing several studies of the benefits and costs to local governments of OCS oil development, a recent analysis concludes that the most reasonable hypothesis is that such developments have no net fiscal impact [12, p. G-62]. This and other evidence suggests that the issue of local infrastructure costs should not be accorded any great weight in determining the future course of OCS development.

SUMMARY AND CONCLUSIONS

We may obtain some idea of the total expected social costs of oil spills from enhanced OCS development, expressed on a per barrel basis, from a recent comprehensive analysis of this issue carried out by the Department of the Interior. This analysis includes estimates of costs in the categories discussed above (and several more, in addition), drawing inferences from a number of well-known studies of the economic costs of oil spills in the marine environment. In its summary of expected costs, the authors take the conservative approach of accepting estimates of

costs which are consistently biased upward, or in favor of the opponents of OCS leasing. Despite this bias, the analysis concludes that the total estimated social costs of enhanced OCS production (net of the benefits gained from reduced levels of oil spillage from tankers which would have had to bring offsetting barrels of imported oil into the U.S.), ranges from $6 million to $53 million per billion barrels of oil equivalent production (BBOE), the amount depending on the particular OCS area being considered [see Table 2].

The analysis above estimated that the social benefits of enhanced OCS oil production were about $25 per barrel in excess of the market value of the oil produced. In comparison to even the highest estimated social cost of OCS production ($53 million per BBOE), this means that the benefit/cost ratio for enhanced OCS oil production is about 500 to 1, or that each $500 in social benefits from enhanced OCS oil production can be expected to create only $1.00 in external social costs. More typically, the benefit/cost ratio will be over 1,000 to 1, as can be seen in Table 2.

In view of this preponderance of benefits over costs, it is unlikely that any reasonable analysis of OCS leasing could dispute the conclusion that enhanced OCS development is in the interest of the nation. Why, then, does social policy toward this issue remain as it is, with the Department of Interior on the defensive and the opponents of OCS leasing continuing to win major political and legal battles?

The answer is suggested by the Interior Department's analysis of the regional distribution of the benefits and costs, which shows that for most affected coastal states, the potential net social costs of OCS development are larger than the benefits which would be paid from federal compensation funds in the case of OCS oil spills [see Table 2]. Even if the distribution of benefits to regions from reduced requirements for federal tax collections were taken account of within the affected regions, several would still end up as losers.[5]

What this means is that for some states the benefit/cost ratio for OCS leasing and development is (or certainly appears to be) less than one, despite the overwhelming preponderance of national benefits. From a public choice perspective, therefore, it is not surprising that politicians representing these states should oppose enhanced OCS leasing near their coastlines. They may or may not recognize the national benefits of OCS leasing, but they will probably believe that their constituents have more to lose than to gain from the proposed new leasing programs.

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[5]See [12], Table 10, p. 65.

Joined in interest with politicians from other affected coastal regions, and using the well-known technique of vote-trading to gain the support of members of Congress from inland states who may have no great concern for the issue one way or the other, opponents of OCS leasing can effectively shackle the federal effort at enhanced leasing. Given the support of local and state politicians, environmental activists who are also opposed to leasing may then use the public hearing process and a sympathetic media to provide a relentlessly negative commentary on the risks of OCS development, with little chance of contradiction except from the promoters of such development--the Department of Interior and the oil industry--who unfortunately have little credibility in the mind of the public.

To break this impasse, the federal government must accept responsibility for a legislative effort to change the distribution of property rights for OCS resources, eventually awarding a share of the bonus and royalty income derived from OCS leases to the states which will bear most of the external social costs associated with OCS development. A blueprint for this approach is suggested by the recently approved legislation awarding coastal states a 27 percent share (amounting to $1.4 billion) of income derived from leases issued within the first three miles of the OCS. The theory underlying this award (that previous state leasing activities had enhanced the value of nearby OCS lands) was clearly questionable, but a political compromise granting a share of OCS revenues to the coastal states was nevertheless reached. A much more convincing body of theory and evidence could be cited in support of the transfer of property rights which is proposed here. Thus, a proportionally greater federal commitment to implement the solution should be forthcoming.

Obviously the federal government will be very reluctant to give up any significant amount of OCS revenue. But the halfway measures currently in place, such as payments made out of OCS funds to all states for the Land and Water Conservation Fund ($800 million per year) and the Historic Preservation Fund ($150 million per year), do nothing to change the incentives of state and local politicians in coastal areas to continue to oppose OCS leasing, because these payments are made irrespective of the risks faced by the states receiving the funds.

One of the keys to bringing this solution about is to recognize that the transfer of property rights being proposed is not simply a "bribe" to recalcitrant states; it is a policy justified by the economic facts in the case. Most importantly, as long as the federal government refuses to share a portion of the OCS pie with the states which now bear a disproportionate part of the social costs of this enterprise, the pace and scope of OCS development will continue to be hobbled, with consequent loss of potential real income for all Americans.

Table 1

LEASING, PRODUCTION, AND REVENUE DATA RELATING TO
OFFSHORE OIL AND GAS LEASES IN THE U.S.

OCS Acreage Offered for Lease 1954-82	175,647,953
OCS Acreage Leased, 1954-82	29,819,834
Total Wells Drilled on OCS Leases, 1954-82	20,956
Cumulative Production of Crude Oil and Condensate from OCS thru 1982 (000 of barrels)	6,027,440
Cumulative Production of Natural Gas from OCS thru 1982 (MCF)	58,182,510,479
Cumulative Production Value of OCS Oil and Gas thru 1982	$105,514,662,534
Bonus, Royalty, and Other Payments to the Federal Government from OCS Leases thru 1982	$58,987,881,911
Payments to the Federal Government from OCS Leases in 1982 Dollars*	$90,351,187,734
Federal Government Share of Total OCS Production Value	56 percent
OCS Oil Production as Share of Total U.S. Oil Production thru 1982	6.82 percent
OCS Gas Production as Share of Total U.S. Gas Production thru 1982	11.59 percent

SOURCE: U.S. Department of the Interior, Minerals Management Service, Federal Offshore Statistics, December 1983.
*Conversion to 1982 dollars made using GNP deflator.

Table 2

ESTIMATED NET DISCOUNTED SOCIAL COSTS ASSOCIATED WITH TOTAL PRODUCTION OF ALL LEASABLE RESOURCES IN EACH OCS AREA (High Range Estimate in Millions of 1987 Dollars)

Region	Total Costs	Net Costs per BBOE[1]
Central Gulf of Mexico	$40.52	$ 9
Western Gulf of Mexico	29.44	6
Navarin Basin	26.97	34
Southern California	20.10	16
Eastern Gulf of Mexico	6.48	14
Northern California	6.36	15
Central California	6.24	15
Beaufort Sea	6.24	21
Chukchi Sea	5.28	13
South Atlantic	4.61	6
Mid-Atlantic	1.51	8
North Atlantic	0.98	22
Washington-Oregon	0.54	10
Straits of Florida	*	53
Gulf of Alaska	*	14
Norton Basin	*	22
Cook Inlet	*	28
St. George Basin	*	18
North Aleutian Basin	*	18
Kodiak	*	15
Shumagin	*	9
Hope Basin	*	8

[1]Per billion barrels of oil equivalent production.
*Less than $500,000.

SOURCE: U.S. Department of Interior, Proposed Program: 5-Year Outer Continental Shelf Oil and Gas Leasing Program for January 1987 - December 1991, Detailed Decision Documents, February 1986, Tables 7 and 8, pp. 45, 47.

REFERENCES

1. American Petroleum Institute, Should Offshore Oil be Put Off Limits, June 1984.

2. Folkerts-Landau, E., "The Social Cost of Imported Oil," The Energy Journal, April 1984, pp. 41-58.

3. Griffin, J. and Steele, H., Energy Economics and Policy, 2d. Ed., Orlando: Academic Press, 1986.

4. Kobrin, P., Production and Revenue Impact of OCS Moratoria, American Petroleum Institute Research Study #033, May 1984.

5. Koons, C., et al., "Environmental Aspects of Produced Water from Oil and Gas Extraction Operations in Offshore Waters", Journal of Petroleum Technology, June 1977, p. 725.

6. Mead, W.J. and Sorensen, P.E., "The Economic Cost of the Santa Barbara Oil Spill," in Santa Barbara Oil Symposium, Marine Science Institute, University of California at Santa Barbara, 1972.

7. Mead, W.J., et al., Offshore Lands: Oil and Gas Leasing and Conservation on the Outer Continental Shelf, San Francisco: Pacific Institute for Public Policy Research, 1986.

8. Plummer, J., "Methods for Measuring the Oil Import Reduction Premium and the Oil Stockpile Premium," The Energy Journal, January 1981.

9. Royal Commission on Environmental Pollution, Eighth Report, Oil Pollution of the Sea, London, October 1981.

10. Sorensen, P.E., "Summary of Damages to the Commonwealth of Puerto Rico Caused by the Bahia Sucia Oil Spill of March 18, 1973, report to the U.S. District Court, P.R., 1977.

11. U.S. Department of Commerce, NOAA, Assessing the Social Costs of Oil Spills: The Amoco Cadiz Case Study, 1983.

12. U.S. Department of Interior, Proposed Program, 5-Year O.C.S. Oil and Gas Leasing Program for January 1987-December 1991, Detailed Decision Documents, February 1986.

13. U.S. Department of Interior, Minerals Management Service, From Policy to Production: Offshore Leasing and Operations, November 1983.

14. U.S. Department of Interior, Minerals Management Service, Federal Offshore Statistics, December 1983.

Session II. Economic Methodologies and Ecological Constraints: Case Studies of Marine Pollution

REGULATION OF MARINE CONTAMINATION UNDER ENVIRONMENTAL UNCERTAINTY: SHELLFISH CONTAMINATION IN CALIFORNIA

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REGULATION OF MARINE CONTAMINATION UNDER ENVIRONMENTAL UNCERTAINTY: SHELLFISH CONTAMINATION IN CALIFORNIA[1]

Contamination of marine resources by industrial, agricultural and municipal wastes is a key environmental concern. Pollution of marine systems impairs both the quantity and quality of services provided. Commercial and sport fish and shellfish harvests may be reduced because of gradual increases in pollution levels or chemical spills. Consumption of fish and shellfish may become more risky because of bioaccumulation of heavy metals and pesticides or because of microbial contamination from sewage and agriculture. Populations of other species may decline, sometimes to the point of extinction. Recreational waters may become unswimmable and recreational areas unusable because of water contamination or oil/chemical spills.

Regulation pertaining to these problems aims at achieving some sort of balance between the value of the activities creating the pollution and that of the marine resources affected. Thus, an assessment of the tradeoffs between polluters' production, the quantity of marine resource output (fish harvests, number of recreation sites, species diversity), and the quality of marine resource output (human health risks from fish or water contamination) is essential for policy determination. One of the central problems in obtaining accurate assessments of these tradeoffs is that there is typically a great deal of uncertainty about the environmental effects. The ways in which contaminants enter and disseminate through the environment tend to be

monitored and modeled only incompletely. Stochastic factors such as weather patterns also tend to have a large degree of influence on them. Many of the fundamental mechanisms underlying physiological responses - both in humans and animals - are not understood well at this time. Moreover, many physiological responses have a multiplicity of causes and contributing factors, making attribution of causality a severe problem.

The general attitude toward these contamination problems emphasizes prevention: policy is increasingly expected to be prospective rather than reactive. This orientation forces regulators to rely on simulation models about whose structure and parameters there is usually considerable uncertainty. Also, the public tends to be quite sensitive to severe outcomes - for example, cancers, birth defects, large wildlife kills or outbreaks of food poisoning - even when these outcomes are rather unlikely. Thus, regulatory decisions need to be based on methodologies which incorporate uncertainty explicitly.

In an earlier paper (Lichtenberg and Zilberman 1985) we proposed a method for constructing uncertainty-adjusted cost curves for environmental risk reduction as a means of meeting this need. This paper applies that approach to a case of human health risk from microbial shellfish contamination. We begin with a discussion of our approach, it uses and its characteristics. We then take up the case of contamination of oyster beds in Tomales Bay, California by dairy runoff. We consider the tradeoff between closing the fishery and requiring the construction of holding ponds to prevent runoff of dairy

wastes as mechanisms for reducing the risk of gastroenteric illness from oyster consumption. We then examine the differences in total cost between standards based on mean risk versus uncertainty-adjusted risk and between cost-efficient requirements for holding pond construction versus requirements which are uniform across dairies. Finally, we consider the implicit value of risk reduction under different regulatory approaches.

Constructing Uncertainty-Adjusted Cost Curves for Environmental Risk Reduction

Our approach to constructing uncertainty-adjusted cost curves for risk reduction essentially involves combining a probabilistic environmental risk assessment - a type of risk assessment that has become increasingly widespread in recent years - with a safety rule decision mechanism. The resulting estimates of uncertainty-compensated tradeoffs between risk and social cost can then be used for policy determination using formal decision criteria (cost-benefit, risk-benefit) or for subjective evaluation of regulatory alternatives.

This procedure has a number of appealing characteristics. First, it makes use of the full range of information available in a practical manner. Even though all of the uncertainty present cannot generally be parameterized, it is important to incorporate as much as possible into regulatory decisions. Second, this approach is more amenable to interdisciplinary cooperation. It utilizes the kinds of information produced by risk analysis and is equivalent to using confidence intervals for statistical

decision making, the method preferred in the natural sciences. Third, because safety rules have been used in a variety of economic applications, they are well understood. Moreover, they have been shown to approximate well expected utility decisions in a number of empirical contexts (Thomson and Hazell 1972). Fourth, the safety rule approach corresponds quite closely to the terms of much of the relevant legislation, which requires regulators to provide adequate protection to public health or the environment within a sufficient margin of safety. It also corresponds to a "disaster-avoidance" approach to decision making which is widespread among the public and the regulatory community. Thus, our approach can be said to suit the preference structure of decision makers.

A set of uncertainty-adjusted cost curves for risk reduction can be derived as follows. Assume that N regulatory instruments $X_1,\ldots,X_N$ are available and that the extent to which each is applied can be measured as an increasing function to social cost. Social cost will include such items as costs of decontaminating the environment, costs of installing exposure reduction devices, welfare losses sustained by consumers and producers due to regulation-induced productivity decreases/cost increases, and government expenditures on monitoring and enforcement. Environmental risk is assumed to be a function of policy, R(X), in a manner defined by a environmental risk assessment. Let R_0 denote a the maximum allowable risk (risk standard) and P represent the frequency with which this standard is met (margin of safety). The regulatory problem can be defined as choosing a

policy mix $X_1^*,\ldots,X_N^*$ to minimize the cost of ensuring that the risk standard R_0 is maintained with a given margin of safety P. This problem can be written formally as

(1) $\min X_1+\ldots+X_N$

subject to the constraint

(2) $\Pr[R(X) \leq R_0] \leq 1-P.$

Repeated application of this procedure letting R_0 vary parametrically and keeping P fixed yields a set of minimum cost tradeoffs between the risk exceeded only with frequency 1-P, denoted R(P), and the total social cost of regulation $X^*=X^*_1+\ldots+X_N^*$ this set amounts to an uncertainty-adjusted cost curve for risk reduction X*[R(P)]. Repeating this procedure over the range of P yields a complete set of such cost curves.

Note that the margin of safety, P, essentially represents the decision maker's aversion to uncertainty: the larger P is, the greater will be the emphasis on the right hand tail of the risk distribution and hence the greater will be the weight placed on uncertainty in choosing a mix of regulatory instruments.

In our earlier paper, we analyzed the optimal mix of regulatory instruments for the risk model proposed for cancer risk assessment by Crouch and Wilson (1981) and subsequently adopted for that purpose by the U.S. Environmental Protection Agency. This model assumes that risk can be expressed well by a multiplicative combination of parameters and that the risk exceeded with any given probability can be represented by a weighted sum of the mean and standard deviation of the risk.[2] We showed that the optimal policy will consist of a portfolio of

measures, some specializing in reducing mean risk and others specializing in reducing uncertainty about risk. Emphasis on reducing uncertainty becomes greater as aversion to uncertainty grows, possibly even to the point where performance with respect to average risk actually declines. Emphasis on mean risk reductions is greater for more toxic contaminants, less controllable risks and for cases where the background uncertainty is large.

The set of optimal policy portfolios for all levels of risk and a given margin of safety P makes up a cost curve for risk reduction with margin of safety P. It is decreasing in risk, i.e., downward sloping in cost-risk space. An increase in toxicity, in uncontrollability or in background uncertainty will increase cost, i.e., push this curve up and to the right. Cost curves associated with higher margins of safety (greater aversion to uncertainty) will lie up and to the right as well.

The slopes of these cost curves, given by the set of lagrange multipliers associated with the constraint (2), represent the marginal cost of reducing risk with a given margin of safety. As such, they give at least a lower bound estimate of social willingness to pay for risk reductions with any given level of aversion to uncertainty. Thus, they can be used to construct a measure of the implicit value of risk reduction with a given margin of safety. In our analysis of the Crouch and Wilson risk model, we showed that this marginal cost of risk reduction increases as aversion to uncertainty increases and as risk, toxicity, uncontrollability and background uncertainty decrease.

These cost curves can be used to generate regulatory decisions using formal methodologies or subjective comparisons of alternatives. For example, cost-benefit solutions can be derived for any given margin of safety by equating marginal benefit and marginal cost; risk benefit solutions can be derived by requiring that the risk/cost ratio be consistent with historical experience (Starr).

Microbial Contamination of Shellfish in Tomales Bay

Pollution has had a severe impact on shellfisheries throughout the U.S. Shellfish harvests have declined significantly because of pollution, overfishing and natural phenomena. Many fisheries have been closed permanently for public health reasons; many other are subjected to recurring temporary closures for the same cause (Larkin and Hunt 1982).

In earlier times, the main public health concern was the spread of typhoid from consumption of sewage-contaminated shellfish. More recently, the focus of concern has shifted to hepatitis A and other viral diseases, although salmonellid gastroenteritis is still a significant problem. Alarm over the health risks from consumption of raw shellfish has heightened in recent months. For example, a recent editorial in the New England Journal of Medicine urged that people cease eating raw shellfish altogether because the risks of gastroenteric illness are simply unacceptable (Du Pont 1986). This sentiment is shared by much of the medical community (Eckholm 1986).

Tomales Bay, located about 50 miles north of San Francisco, houses a small cultivated oyster fishery. The main threat to the fishery comes from effluent runoff from the surrounding dairy industry. Under normal conditions, runoff from the manure spread over each dairy's disposal area will have a negligible effect on water quality and hence on the fishery. During severe rainstorms, though, the manure will be washed into the watershed. The resulting fecal contamination of the bay will lead to contamination of the oyster beds. If the water quality around the oyster beds fails to meet public health standards, the fishery will be closed. Current standards, expressed in terms of the most probable number (MPN) of bacteria, are median values of 70 MPN/100 ml water for total coliform bacteria and 14 MPN/100 ml water for fecal coliform bacteria (Hunt 1977).

Contamination can be reduced by constructing holding ponds at the dairies. Each pond will retain effluent up to a fixed capacity. Because the manure floats, it will be washed into the watershed as soon as that capacity is exceeded. The severity and frequency of pollution will thus depend on the capacities of the holding ponds built. Current regulations require all dairies to maintain facilities to hold a 10-year 24-hour rainfall in addition a maximum average rainfall of 5.75 inches occurring in the previous three weeks (Rafter et al. 1974).

Hochman, Zilberman and Just (1977) analyzed optimal regulation of the Tomales Basin under the assumptions of zero acceptable risk and of uniformity of holding ponds capacities across dairies. In this case pollution is all-or-nothing: either

the bay is contaminated and the fishery shut down or the bay remains clean and the fishery stays open. Their analysis suggested that, to maximize the sum of fishery and dairy profits, dairies should be required to construct ponds capable of holding 50-year 24-hour rainfall events so that the fishery would be closed only one-fifth as often as under current regulations.

This characterization of the situation has several shortcomings. First, it ignores heterogeneity among dairies. Because of topoographical differences, the costs of constructing a holding pond of any given capacity tend to vary widely; Hochman, Zilberman and Just's data indicate that they may differ by a factor of four or more. This suggests that any given standard can be achieved at a lower overall cost by discriminating among dairies, i.e., requiring dairies in more favorable locations to build large holding ponds and those in less favorable locations to build smaller ones. Second, it examines only a simple, very stringent risk standard - essentially one of zero acceptable risk. This implies that the fishery must be closed whenever pollution is present. Thus, the analysis ignores the possibilities of substitution between closing the fishery and building holding ponds, which may permit standards to be achieved at lower cost. Moreover, it would be useful to examine the choice of a standard as well. For this purpose, it would be preferable to generate costs for a range of standards. Third, because the risk is all-or-nothing, uncertainty enters in a very restricted way, namely as the probability that a risk occurs. Examination of a broader range

of standards would have to incorporate much more of the probability distribution characterizing the risk. Fourth, only one source of uncertainty - rainfall - is modeled. There is also significant uncertainty about other factors affecting risk, including the amounts of dairy waste present, the limits between runoff and microbial contamination of the bay, the links between water quality and shellfish contamination and dose-response relations for illness resulting from the consumption of contaminated shellfish.

This study attempts to overcome some of these shortcomings. In what follows, we combine Hochman, Zilberman and Just's data on runoff control costs with a model of acute gastroenteritis risk from raw oyster consumption. We employ the approach described in the previous section to derive sets of uncertainty-adjusted, cost-efficient tradeoffs between total control cost and risk which we then use to analyze alternative regulatory approaches.

Risk of Acute Gastroenteritis from Microbial Shellfish Contamination

For simplicity, the risk of acute gastroenteritis from microbial shellfish contamination was modeled as a multiplicative combination of variables reflecting water quality, microbial uptake by oysters, and human physiological response. Two policy instruments were included: (1) a requirement that each dairy build a holding pond of a designated size, not necessarily identical, and (2) closure of the fishery.

Each cow was assumed to produce an identical constant amount of manure and each pound of manure was assumed to have a constant, identical microbial content. The microbial content of manure was assumed to be measured, accurately by the coliform count. Manure in runoff was assumed to be mixed uniformly in the bay waters around the oyster beds. This is undoubtedly an oversimplification, but does constitute a reasonable first approximation (see for example the data presented by the Northeast Technical Services Unit 1980).

Under these assumptions, water quality around the oyster beds will be proportional to the number of cows generating effluent. Runoff is affected by two things: (1) the sizes of holding ponds at the dairies, which is a function of policy, denoted X_1 and (2) rainfall, denoted R. Write the size distribution of holding ponds (that is, the number of dairies having holding ponds of capacity k) as $W(k, X_1)$ and the set of dairies generating runoff for any given 24-hour rainfall level r as $B[W(k,X_1),R]$. If there are J dairies having herds of sizes $H_1,\ldots,H_J$, then water quality can be written as $q\sum_{j\varepsilon B} H_j$, where q is a constant of proportionality.

To estimate q, it was assumed that during extremely heavy rainstorms effluent runs off at all dairies and that this effluent is the sole source of microbial contamination as measured by the fecal coliform count. This ignores the contributions from human sources such as defective septic tanks, which are important in certain areas but of secondary significance overall. The maximum fecal coliform count measured

over a broad range of oyster beds was 16,000 MPN/100 ml water (Northeast Technical Services Unit 1980). The total number of cows at dairies in the watershed, as reported by Hochman, Zilberman and Just was 13,200, implying a value of q = 1.2 MPN/100 ml water per cow.

Uptake of microbial contaminants by the oyster population was assumed to be proportional to water quality around the oyster beds. Letting u denote this uptake parameter, the proportion of oysters contaminated can be written as $uq \sum_{j \in B} H_j$. This parameter was estimated using data presented by Andrews et al. (1975) in their study of fecal coliform counts as an indicator of bacteriological contamination. They reported percentages of oyster samples testing positive for salmonella for five ranges of fecal coliform counts. These percentages were regressed on the midpoints of the fecal coliform count ranges without a constant term, with an additional assumption that the maximum observed fecal coliform count was 16,000 MPN/100 ml water, to obtain a estimate of u = 0.000057. The standard deviation of this estimate was 0.000022.

This approach has several shortcomings. First, it assumes that the presence of salmonella in oyster meat indicates a sufficient level of microbial contamination to cause illness. It would be preferable to measure the extent to which the meat is contaminated as well; however, the literature on dose-response relations looks only at whether or not the shellfish are contaminated, so this additional level of detail would add little of substance to the analysis. More significant is the implicit

assumption that the presence of salmonella is also a good indicator of viral contamination. The results of a number of studies indicate the contrary, specifically that the fecal coliform count in surrounding waters tends not to be such a good indicator of viral contamination (see for instance Gerba et al. 1980). However, no better indicator seems to be available at present.

A number of studies have shown that bacterial and viral populations remain relatively constant after processing and cold storage for several weeks (Kelley and Arcisz 1954; Wilson and McCleskey 1951; Erickson, Vasconcelos and Presnell 1967; Di Girolamo, Liston and Matches 1970; Hood, Baker and Singleton 1984). Because this study is concerned with incremental risk from water contamination, additional contamination from shucking and handling and improper storage were ignored.

The probability of contracting gastroenteritis from eating contaminated oysters was assumed to be proportional to the proportion of oysters contaminated. Letting d represent the dose-response parameter, this probability can be written as $duq \sum_{j \epsilon B} H_j$. Variations in this dose-response relation due to variations in the number of oysters eaten or the extent of contamination of each oyster were ignored. No data on bacteriological poisonings from consuming contaminated shellfish were known to us, so epidemiological studies of viral infections were used exclusively. Several such studies investigated incidents where guests at large parties had been served contaminated shellfish. The proportion of people consuming these

shellfish who contracted gastroenteritis ranged from .36 to .65 (Gill et al. 1983; Gunn et al. 1982; Morse et al. 1986). The dose-response parameter d was given a value .45, in accord with the incidence of gastroenteritis reported by Gunn et al. and the combined incidence of gastroenteritis and hepatitis reported by Morse et al.

Finally, the risk of gastroenteritis may be eradicated by closing the fishery and preventing harvest of the shellfish until they have decontaminated themselves. This effect can be represented by a function $o(X_2)$ which takes on a value of one when the fishery remains open and zero when the fishery is closed.

The risk of acute gastroenteritis from consumption of contaminated oysters, denoted t, can thus be written as $t = o(X_2)duq\sum_{j\varepsilon B} H_j$, a function of policies relating to the fishery, X_2, and to the construction of holding ponds, X_1.

As a first step, we will consider only one source of uncertainty, namely the randomness of 24-hour rainfall. Other sources of uncertainty will be incorporated in a future study. The probability distribution of 24-hour rainfall was modeled as a Pearson type II distribution as in Hochman, Zilberman and Just. Using data from the California Department of Water Resources (1974) they estimated that the 24-hour rainfall exceeded with probability b was 2.928 + K(b)0.413, where K(b) was chosen to correspond to a Pearson type III distribution with a skrewness of 1.3. Letting F(r) denote the probability that 24-hour rainfall R does not exceed a given level r, the risk associated with any given level of 24-hour rainfall can be written:

$$t = [1-F(r)]o(X_2)duq \sum_{j \varepsilon B[W(k,X_1),r]} H_j .$$

Calculation of Cost Curves for Gastroenteric Risk Reduction

Numerical methods were used to estimate total social costs and probability distributions of risk under alternative policies and to determine the minimum cost policy mix for each level of uncertainty-adjusted risk. Three levels of aversion to uncertainty were analyzed: neutrality, which implies regulation based on mean risk, and safety margins of 95% and 99%.

The Tomales Bay fishery produces a small share of total oyster production in the San Francisco Bay area; loss of the harvest was thus assumed to have a negligible impact on prices. The analysis performed by Hochman, Zilberman and Just suggested that holding pond construction costs would never be great enough to force dairies out of business, so this potential impact was ignored. It was assumed also that periodic closure of the fishery would have negligible long run impacts on production or employment. This characterization seems reasonable in light of the fact that most growers are small, individual operators using a minimum of equipment, so that entry is quite inexpensive. Finally, it was assumed that monitoring and enforcement costs are negligible, which also seems reasonable in light of the fact that the California Department of Health Services has only one person responsible for shellfish in all of Northern California. These assumptions imply that total expected social cost is closely approximated by the sum of amotized holding pond construction costs and expected losses in revenue from closure of the fishery.

The probability distribution of 24-hour rainfall events was partitioned into a number of discrete segments. For each of these states of nature the data collected by Hochman, Zilberman and Just were used to estimate the cost at each dairy of building a holding pond with capacity sufficient to hold no more than the corresponding 24-hour rainfall given the average maximum rainfall of 5.75 inches in the preceding three weeks. Following Hochman, Zilberman and Just, total construction costs were annualized by multiplying them by an interest rate of 10%. The result of these calculations was a set of functions for each dairy $C_1(b),\ldots,C_J(b)$ representing the annualized cost of building a holding pond with sufficient capacity for 24-hour rainfalls exceeded only 100b% of the time.

Expected revenue losses from closure of the fishery were estimated by multiplying the frequency of closures under alternative policies times the annual revenue of the fishery. Following Hochman, Zilberman and Just, fishery revenue was estimated at $500,000 per year. No adjustment for inflation was made in either case; thus, all costs are expressed in terms of 1973 dollars.

Costs and risks under a policy approach relying only on the construction of holding ponds under neutrality toward uncertainty were estimated as follows. Each dairy was assumed to behave as if it faced a tax T on effluent (effectively, on herd size). Constructin a holding pond with a capacity exceeded only with probability b will cost $C_j(b)$. The expected tax payments in this case will amount to bTH_j. Minimization of the sum of these two

costs implies that the optimal capacity for the j^{th} dairy, b^*_j, should be set to equate the marginal construction cost $C'_j(b^*_j)$ with the maximum tax liability TH_j. Construction cost for the j^{th} dairy is then $C_j(b^*_j)$ and total construction cost is found by summing up construction costs over all dairies. The choice of a capacity also gives a two-point distribution of effluent: effluent is zero with probability $1-b^*_j$ and proportional to H_j with probability b^*_j. Aggregation over the full set of dairies gives a probability distribution of total effluent which was then transformed into a probability distribution of risk using the model described in the previous section. Mean risk was calculated from that probability distribution, for each tax level, giving a risk, cost pair. Repeating this procedure over a range of values of T gave a set of such pairs, i.e. a cost curve for risk reduction.

For policies based on 95% or 99% margins of safety, the same procedure was followed with the addition of a constraint that holding pond capacity never be large enough for 24-hour rainfalls exceeded less frequently than the margin equal to P, this meant $b^*_j \leq 1-P$. The measure of risk used under these policies was the level exceeded only 100(1-P)% of the time, 5% and 1% for the 95% and 99% margins of safety, respectively.

The combined impact of the two policies of building ponds and closing the fishery was estimated using the same procedures with the additional constraint that pond capacity never exceed the frequency with which the fishery would be closed. Thus, if the closure policy were to prevent harvest after 24-hour

rainfalls no less than 100-yearevents, dairies were constrained to build ponds with capacities no greater than 100-year 24-hour rainfalls. If the fishery were closed with probability c, this implied a constraint of $b^*_j \leq c$ for neutrality with respect to uncertainty and $b^*_j \leq 1-P+c$ for a margin of safety P. The total cost of these policies was found by adding expected revenue losses to total construction costs as described above.

The efficient mix of policies and minimum-cost tradeoff curves for each level of aversion to uncertainty were estimated by comparing the total cost curves generated by these different closure rules. The resulting social cost curves for risk reductions were the lower envelopes of the total cost curves for each of these rules.

For policies under neutrality with respect to uncertainty, the cost of achieving a mean risk standard using uniform holding pond construction requirements was estimated by finding the frequency with which the maximum risk level would have to occur to generate the same average risk. Letting t* denote this maximum risk level, the capacity required to achieve any level of average risk t is $b^*u = t/t^* + c$, where c is the probability that the fishery is closed. Total construction costs for this uniform capacity bu* and revenue losses from closure were then calculated and compared as described above.

Such fine tuning is not possible for policies under 95% and 99% margins of safety: risk is either at a maximum or zero. The key problem in such cases is calculating the minimum cost combination of closure and construction required to achieve zero

risk. This was done by comparing the total cost curves for alternative closure policies using uniform building requirements.

Empirical Results

Some preliminary results of this analysis are shown in Figures 1-4.

Consider first the choice of policy instruments between closing the fishery and requiring construction of holding ponds. As Figure 1 shows, closing the fishery as seldom as 1% of the time is suboptimal except for very low risk standards. This occurs because closing the fishery is a relatively expensive way to reduce risk. It costs $5000 to reduce risk by 1% by closing the fishery. Building holding ponds becomes that costly only at very large capacities. For example, Hochman, Zilberman and Just found that, under a uniform building requirement, the marginal costs of these two instruments were equal only when holding ponds were capable of holding up to 50-year 24-hour rainfalls. The buildining requirements derived here are even less costly, implying that the marginal costs of the two instruments will be equated only at even larger capacities. Expressed in terms of risk, Figure 1 indicates that closing the fishery for 100-year and larger 24-hour rainfalls in cost-efficient under neutrality toward uncertainty only for risk standards of .001 and less.

The optimal policy mix changes as aversion to uncertainty increases. Figures 2 and 3 compare fishery closure with efficient building requirements under margins of safety for 95% and 99%, respectively. Closure is not used under a 95% safety

margin. This occurs because 24-hour rainfalls larger than 20-year events are of no concern to policy makers at this level of aversion to uncertainty. As we have seen, fishery closure becomes cost-competitive with building requirements only at much larger capacities. In contrast, closure is under for much higher risk standards under a 99% safety margin than under neutrality toward uncertainty. This occurs because very large, infrequent 24-hour rainfalls count very heavily under this level of aversion to uncertainty.

Now consider the impact of aversion to uncertainty on the cost curve for risk reduction. There are two exceptions to this. First, the 95% and 99% margin of safety cost curves are identical at maximum risk. This occurs because the construction of holding ponds has no effect on risk with these margins of safety until capacities of 20-year and 100-year 24-hour rainfalls are reached. Second, mean risk standards becomes more expensive than 95% margin of safety standard at very low risk levels. The reason for this is that decisions made under a 95% safety rule ignore risks that occur less frequently than 5% of the time, i.e., when very large rainstorms occur. Thus, a "zero" risk level is attained even though some residual risk actually occurs (albeit infrequently). It is precisely these residual risks which are the most expensive to eradicate. Under mean risk oriented regulation, their share of weight in decision making grows as the overall risk level declines, so that regulation becomes increasingly costly. These low-frequency risks do play an important part regulation with a 99% margin of safety, though,

with the result that the cost of mean risk standards never exceeds the cost of 99% margin of safety standard.

Excluding these two cases, it can be seen that the premium paid for reductions in uncertainty declines as the risk standard decreases. For example, achieving a 99% margin of safety costs about $10,000 more than achieving a 95% margin of safety for higher risk levels. This implies an average "uncertainty premium" of about $2500 for each 1% increase in the margin of safety. For very low levels of risk, though, this difference decreases. The gap between cost under neutrality toward uncertainty and under a 95% margin of safety is even greater for higher risk levels, about $80,000 for a risk of 0.35, for example. It decreases much more rapidly, though, and declines to about $5000 for very low risk levels. Neutrality toward uncertainty in this model corresponds to a safety margin of 45%, implying that the "uncertainty premium" averages about $1,600 for each 1% increase in the margin of safety for very high risk levels and falls as low as $100 for very low risk levels. This suggests that the cost of aversion to uncertainty increases as the level of aversion to uncertainty rises, a result consistent with the standard literature on risk and uncertainty.

Finally, consider the increase in social cost from the use of uniform holding pond size requirements such as are now in force. Preliminary calculations suggest that the gap between the minimum cost of risk reduction and cost under a uniform building policy is greatest at high risk for all levels of aversion to uncertainty and that the two converge as the risk standard

approaches zero. The latter result occurs because, to attain a zero risk standard, it is generally necessary to require holding ponds of the maximum relevant capacity, effect a uniform building requirement. Under neutrality with respect to uncertainty, the additional social cost of a uniform policy is about $55,000 at high risk levels, declines rather rapidly to $2,000 as risk declines and disappears as risk approaches zero. For safety margins of 95% and 99%, this additional cost reaches only about $7,500 and $5,000 respectively, suggesting that efficiency losses decline as aversion to uncertainty increases.

Concluding Remarks

We have argued that uncertainty tends to matter a great deal in environmental policy problems - including those concerning marine pollution - because of the incompleteness of the knowledge base from which decisions must be made and because of the expressed preferences of the general public. This implies in turn that formal methods for making environmental policy decisions should incorporate uncertainty explicitly. We have proposed once such methodology, which applies a safety rule decision mechanism to a probablistic model of environmental risk to generate uncertainty-adjusted cost curves for risk reduction. We argued that this approach combines several theoretically appealing features and practical usefulness.

The empirical analysis of microbial contamination of oysters in Tomales Bay by dairy wastes bears out these contentions. Calculation of the uncertainty-adjusted cost curves was feasible

and the curves behaved as expected: they were decreasing in risk and increasing in aversion to uncertainty. Aversion to uncertainty was shown to have a significant influence on the choice of policy instruments: closing the fishery turned out to be cost-efficient only for very high levels of aversion to uncertainty.

The empirical example also demonstrated some interesting characteristics of decisions made in this fashion. The premium in social cost required to achieve reductions in risk and increasing in aversion to uncertainty.

Regarding pollution control measures, the analysis suggested that inefficiencies due to the imposition to uniform control technologies tend to be greatest for high risk levels and low aversion to uncertainty. This result has some interesting implications for broader pollution control policies.

In sum, we believe that the analysis shows that our proposed methodology can be useful tool for environmental decision making and yields some interesting perspectives on a number of aspects of environmental policy questions.

Footnotes

[1]Although the information described in this article has been funded wholly or in part by the United States Environmental Protection Agency under assistance agreement CR811200-03 to the Western Consortium for the Health Professions, it has not been subjected to the Agency's required peer and administrative review and therefore does not necessarily reflect the views of the Agency and no official endorsement should be inferred. We would like to thank Dave Alton, Doug Price, Bob Hultquist and Mel Yee for their assistance. Responsibility for any errors is, of course, ours.

[2]This assumption is not very restrictive. Such representations can be derived for any probability distribution using Chebyshev's inequality or the partial moment methods proposed by Berck and Hihm (198?) and by Atwood.

References

Erik Lichtenberg and David Zilberman, "Efficient Regulation of Environmental Health Risks", Western Consortium for the Health Professions, San Francisco, CA, 1985.

Charincy Starr, Science.

Eithan Hochman, David Zilberman and Richard Just, "Internalization in a Stochastic Pollution Model", Water Resources Research 13:877-881, 1977.

Dale Morse, et al., "Widespread Outbreaks of Clam- and Oyster-Associated Gastroenteritis", New England Journal of Medicine 314:678-681, 1986.

O.N. Gill et al., "Epidemic of Gastroenteritis Caused by Oysters Contaminated with Small Round Structured Viruses", British Medical Journal 287:1532-1534, 1983.

Robert A. Gunn et al., "Norwalk Virus Gastroenteritis Following Raw Oyster Consumption", American Journal of Epidemiology 115:348-351, 1982.

R. Di Girolamo, J. Liston and J.R. Matches, "Survival of Virus in Chilled, Frozen and Processed Oysters", Applied Microbiology 20:58-63, 1970.

W.H. Andrews et al., "Comparative Validity of Members of the Total Coliform and Fecal Coliform Groups for Indicating the Presence of Salmonella in the Eastern Oyster, Crassostrea virginica", Journal of Milk and Food Technology 38:453-456, 1975.

M.A. Hood, R.M. Baker and F.L. Singleton, "Effect of Processing and Storing Oyster meats on Concentrations of Indicator Bacteria, Vibrios and Aeromonas hydrophila", Journal of Food Protection 47:598-601, 1984.

J. Erickson, G.J. Vasconcelos and M.W. Presnell, "Time-Temperature Effects on the Bacteriological Quality of Stored Shellfish", _____ 32:121-124, 1967.

Thomas E. Wilson, and C.S. McCleskey, "The Effect of Storage on Indicator Organisms in Shucked Oysters and a Comparison of Methods for the Determination of Enterococci", Food Research 16:377-382, 1951.

C.B. Kelly and William Arcisz, "Survival of Enteric Organisms in Shellfish", Public Health Reports 69:1205-1210, 1954.

Northeast Technical Services Unit, "Sanitary Survey of Shellfish Waters, Tomales Bay, California, February and March 1980", Davisville, RI, 1980.

Edward P. Larkin and Daniel A. Hunt, "Bivalue Mollusks: Control of Microbiological Contaminants", BioScience 32:193-197, 1982.

Herbert L. DuPont, "Consumption of Raw Shellfish - is the Risk Now Unacceptable?", New England Journal of Medicine 314:707-708, 1986.

Erik Eckholm, "Warnings Issued in Eating Shellfish that is Uncooked", New York Times, March 13, 1986.

R. Rafter et al., "Resource Conservation Data: Dairy Pollution Control Costs in Marin County, Minimum Guidelines for Protection of Water Quality from Animal Waste, Final Report", U.S. Soil Conservation Service, Marin County, CA, 1974.

California Department of Water Resources, "Summary of Short-Duration Precipitation Frequency in the San Francisco Bay Area", Sacramento, 1974.

C.P. Gerba, et al., "Bacterial Indicators and Environmental Factors as Related to Contamination of Oysters by Enteroviruses", Journal of Food Protection 43:99-101, 1979.

D.A. Hunt, "Indicators of Quality for Shellfish Waters" in A.W. Hoadley and B.J. Dutka (eds.), Bacterial Indicators/Health Hazards Associated with Water ASTM STO 635, 1977.

K.J. Thomson and P.B.R. Hazell, "Reliability of Using the Mean Absolute Deviation to Derive Efficient E, V Farm Plans", American Journal of Agricultural Economics 54:503-506, 1972.

E. Crouch and R. Wilson, "Regulation of Carcinogens", Risk Analysis 1:47-57, 1981.

Figure 1
Construction vs. Closure: Mean Risk

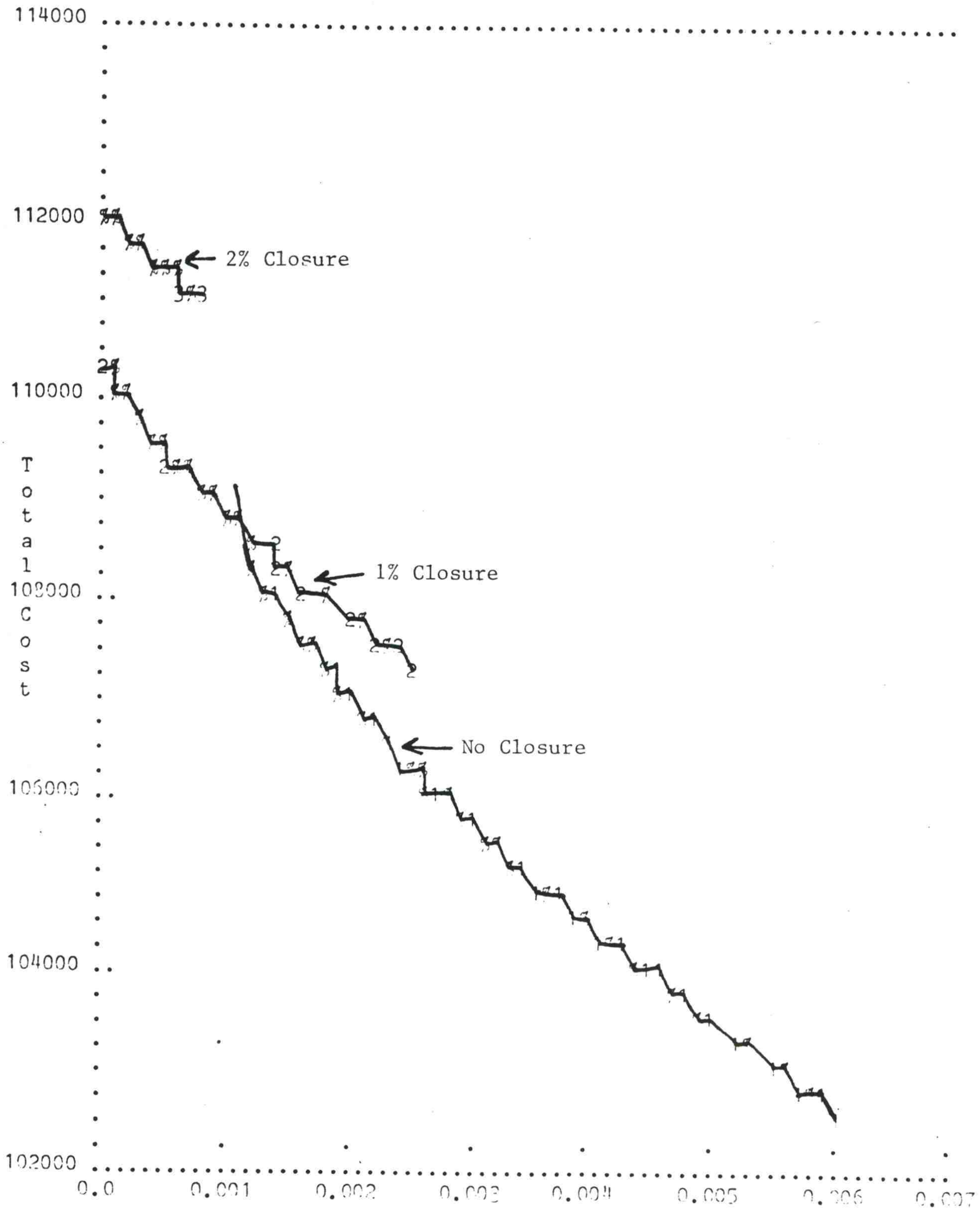


Figure 2

Construction vs. Closure: 95% Safety Margin

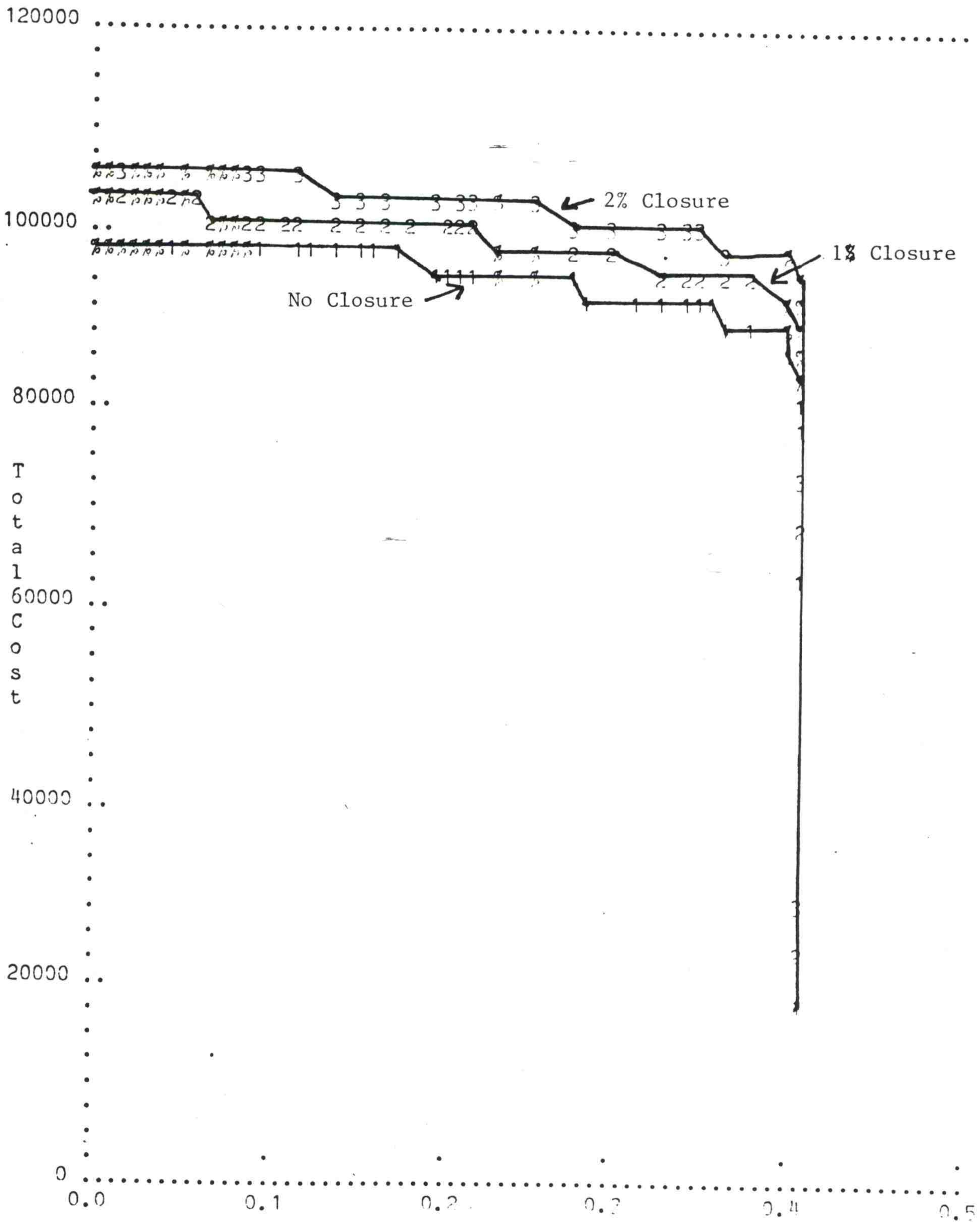
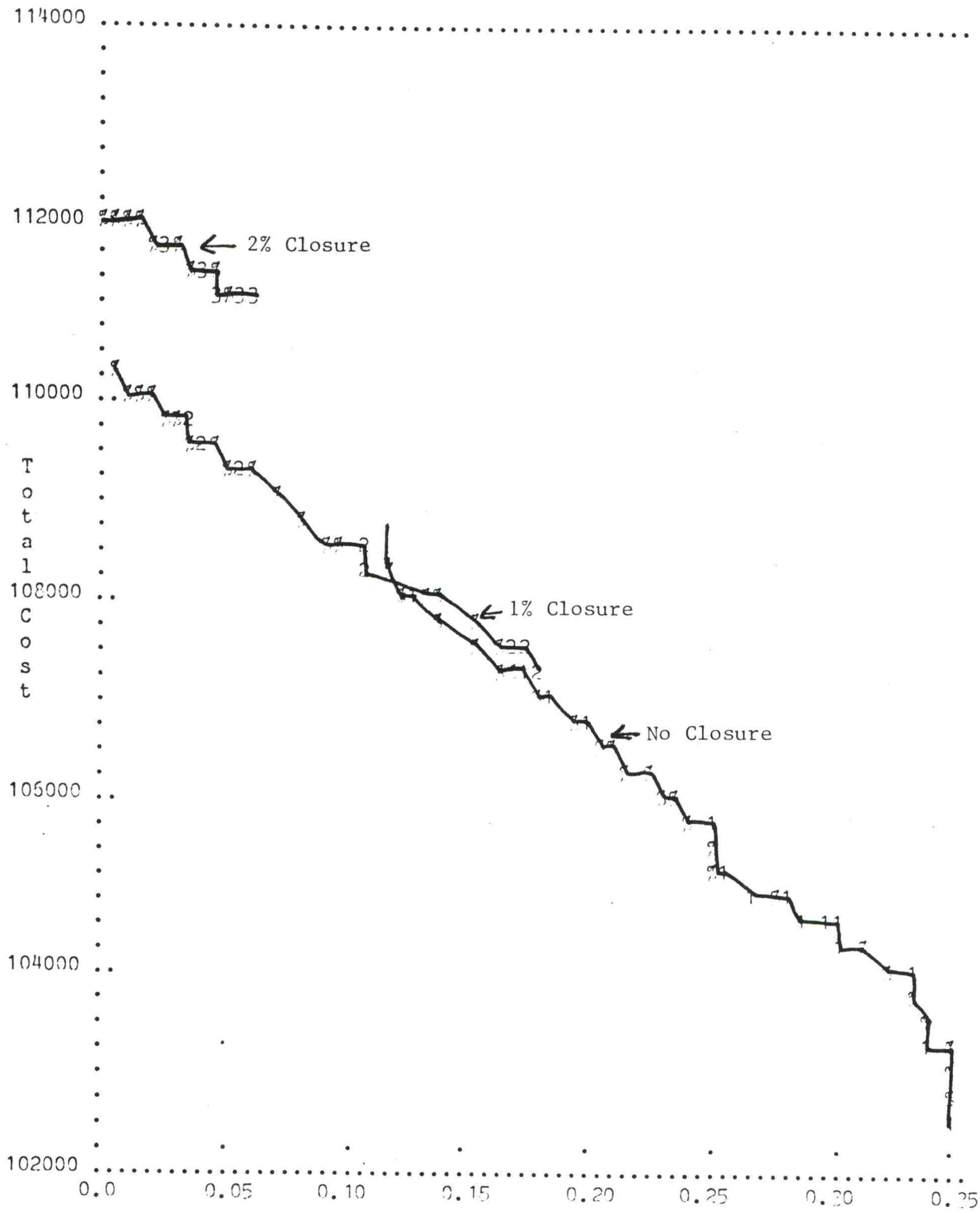


Figure 3

Construction vs. Closure: 99% Safety Margin

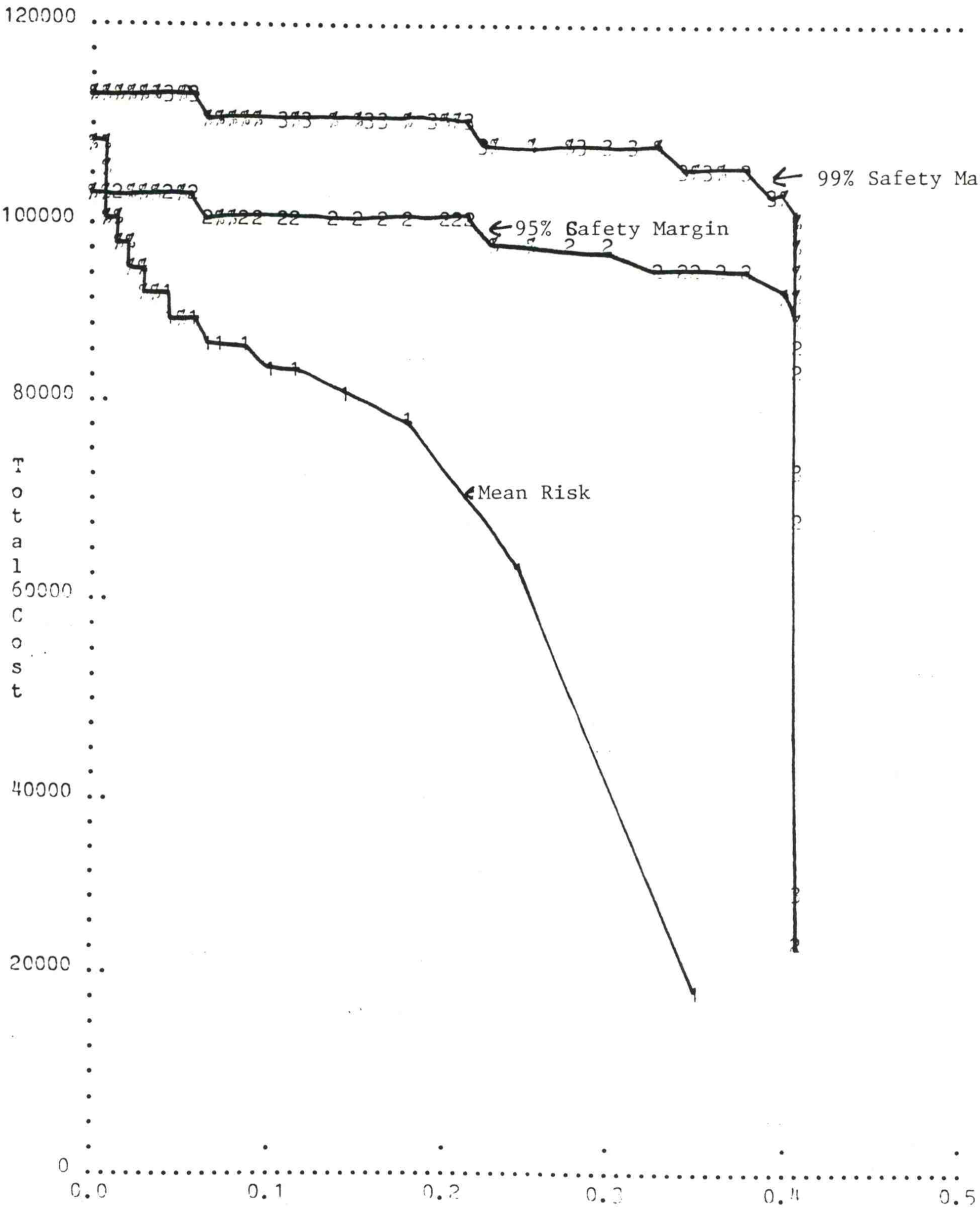


99% Risk

1=No Closure, 2=1% Closure, 3=2% Closure

Figure 4

Impacts of Aversion to Uncertainty



REGULATION OF MARINE CONTAMINATION UNDER ENVIRONMENTAL UNCERTAINTY: SHELLFISH CONTAMINATION IN CALIFORNIA

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REGULATION OF MARINE CONTAMINATION UNDER ENVIRONMENTAL UNCERTAINTY: SHELLFISH CONTAMINATION IN CALIFORNIA[1]

Contamination of marine resources by industrial, agricultural and municipal wastes is a key environmental concern. Pollution of marine systems impairs both the quantity and quality of services provided. Commercial and sport fish and shellfish harvests may be reduced because of gradual increases in pollution levels or chemical spills. Consumption of fish and shellfish may become more risky because of bioaccumulation of heavy metals and pesticides or because of microbial contamination from sewage and agriculture. Populations of other species may decline, sometimes to the point of extinction. Recreational waters may become unswimmable and recreational areas unusable because of water contamination or oil/chemical spills.

Regulation pertaining to these problems aims at achieving some sort of balance between the value of the activities creating the pollution and that of the marine resources affected. Thus, an assessment of the tradeoffs between polluters' production, the quantity of marine resource output (fish harvests, number of recreation sites, species diversity), and the quality of marine resource output (human health risks from fish or water contamination) is essential for policy determination. One of the central problems in obtaining accurate assessments of these tradeoffs is that there is typically a great deal of uncertainty about the environmental effects. The ways in which contaminants enter and disseminate through the environment tend to be

monitored and modeled only incompletely. Stochastic factors such as weather patterns also tend to have a large degree of influence on them. Many of the fundamental mechanisms underlying physiological responses - both in humans and animals - are not understood well at this time. Moreover, many physiological responses have a multiplicity of causes and contributing factors, making attribution of causality a severe problem.

The general attitude toward these contamination problems emphasizes prevention: policy is increasingly expected to be prospective rather than reactive. This orientation forces regulators to rely on simulation models about whose structure and parameters there is usually considerable uncertainty. Also, the public tends to be quite sensitive to severe outcomes - for example, cancers, birth defects, large wildlife kills or outbreaks of food poisoning - even when these outcomes are rather unlikely. Thus, regulatory decisions need to be based on methodologies which incorporate uncertainty explicitly.

In an earlier paper (Lichtenberg and Zilberman 1985) we proposed a method for constructing uncertainty-adjusted cost curves for environmental risk reduction as a means of meeting this need. This paper applies that approach to a case of human health risk from microbial shellfish contamination. We begin with a discussion of our approach, it uses and its characteristics. We then take up the case of contamination of oyster beds in Tomales Bay, California by dairy runoff. We consider the tradeoff between closing the fishery and requiring the construction of holding ponds to prevent runoff of dairy

wastes as mechanisms for reducing the risk of gastroenteric illness from oyster consumption. We then examine the differences in total cost between standards based on mean risk versus uncertainty-adjusted risk and between cost-efficient requirements for holding pond construction versus requirements which are uniform across dairies. Finally, we consider the implicit value of risk reduction under different regulatory approaches.

Constructing Uncertainty-Adjusted Cost Curves for Environmental Risk Reduction

Our approach to constructing uncertainty-adjusted cost curves for risk reduction essentially involves combining a probabilistic environmental risk assessment - a type of risk assessment that has become increasingly widespread in recent years - with a safety rule decision mechanism. The resulting estimates of uncertainty-compensated tradeoffs between risk and social cost can then be used for policy determination using formal decision criteria (cost-benefit, risk-benefit) or for subjective evaluation of regulatory alternatives.

This procedure has a number of appealing characteristics. First, it makes use of the full range of information available in a practical manner. Even though all of the uncertainty present cannot generally be parameterized, it is important to incorporate as much as possible into regulatory decisions. Second, this approach is more amenable to interdisciplinary cooperation. It utilizes the kinds of information produced by risk analysis and is equivalent to using confidence intervals for statistical

decision making, the method preferred in the natural sciences. Third, because safety rules have been used in a variety of economic applications, they are well understood. Moreover, they have been shown to approximate well expected utility decisions in a number of empirical contexts (Thomson and Hazell 1972). Fourth, the safety rule approach corresponds quite closely to the terms of much of the relevant legislation, which requires regulators to provide adequate protection to public health or the environment within a sufficient margin of safety. It also corresponds to a "disaster-avoidance" approach to decision making which is widespread among the public and the regulatory community. Thus, our approach can be said to suit the preference structure of decision makers.

A set of uncertainty-adjusted cost curves for risk reduction can be derived as follows. Assume that N regulatory instruments $X_1,\ldots,X_N$ are available and that the extent to which each is applied can be measured as an increasing function to social cost. Social cost will include such items as costs of decontaminating the environment, costs of installing exposure reduction devices, welfare losses sustained by consumers and producers due to regulation-induced productivity decreases/cost increases, and government expenditures on monitoring and enforcement. Environmental risk is assumed to be a function of policy, R(X), in a manner defined by a environmental risk assessment. Let R_0 denote a the maximum allowable risk (risk standard) and P represent the frequency with which this standard is met (margin of safety). The regulatory problem can be defined as choosing a

policy mix $X_1^*,\ldots,X_N^*$ to minimize the cost of ensuring that the risk standard R_0 is maintained with a given margin of safety P. This problem can be written formally as

(1) $\min X_1+\ldots+X_N$

subject to the constraint

(2) $Pr[R(X) \leq R_0] \leq 1-P.$

Repeated application of this procedure letting R_0 vary parametrically and keeping P fixed yields a set of minimum cost tradeoffs between the risk exceeded only with frequency 1-P, denoted R(P), and the total social cost of regulation $X^*=X^*_1+\ldots+X_N^*$ this set amounts to an uncertainty-adjusted cost curve for risk reduction $X^*[R(P)]$. Repeating this procedure over the range of P yields a complete set of such cost curves.

Note that the margin of safety, P, essentially represents the decision maker's aversion to uncertainty: the larger P is, the greater will be the emphasis on the right hand tail of the risk distribution and hence the greater will be the weight placed on uncertainty in choosing a mix of regulatory instruments.

In our earlier paper, we analyzed the optimal mix of regulatory instruments for the risk model proposed for cancer risk assessment by Crouch and Wilson (1981) and subsequently adopted for that purpose by the U.S. Environmental Protection Agency. This model assumes that risk can be expressed well by a multiplicative combination of parameters and that the risk exceeded with any given probability can be represented by a weighted sum of the mean and standard deviation of the risk.[2] We showed that the optimal policy will consist of a portfolio of

measures, some specializing in reducing mean risk and others specializing in reducing uncertainty about risk. Emphasis on reducing uncertainty becomes greater as aversion to uncertainty grows, possibly even to the point where performance with respect to average risk actually declines. Emphasis on mean risk reductions is greater for more toxic contaminants, less controllable risks and for cases where the background uncertainty is large.

The set of optimal policy portfolios for all levels of risk and a given margin of safety P makes up a cost curve for risk reduction with margin of safety P. It is decreasing in risk, i.e., downward sloping in cost-risk space. An increase in toxicity, in uncontrollability or in background uncertainty will increase cost, i.e., push this curve up and to the right. Cost curves associated with higher margins of safety (greater aversion to uncertainty) will lie up and to the right as well.

The slopes of these cost curves, given by the set of lagrange multipliers associated with the constraint (2), represent the marginal cost of reducing risk with a given margin of safety. As such, they give at least a lower bound estimate of social willingness to pay for risk reductions with any given level of aversion to uncertainty. Thus, they can be used to construct a measure of the implicit value of risk reduction with a given margin of safety. In our analysis of the Crouch and Wilson risk model, we showed that this marginal cost of risk reduction increases as aversion to uncertainty increases and as risk, toxicity, uncontrollability and background uncertainty decrease.

These cost curves can be used to generate regulatory decisions using formal methodologies or subjective comparisons of alternatives. For example, cost-benefit solutions can be derived for any given margin of safety by equating marginal benefit and marginal cost; risk benefit solutions can be derived by requiring that the risk/cost ratio be consistent with historical experience (Starr).

Microbial Contamination of Shellfish in Tomales Bay

Pollution has had a severe impact on shellfisheries throughout the U.S. Shellfish harvests have declined significantly because of pollution, overfishing and natural phenomena. Many fisheries have been closed permanently for public health reasons; many other are subjected to recurring temporary closures for the same cause (Larkin and Hunt 1982).

In earlier times, the main public health concern was the spread of typhoid from consumption of sewage-contaminated shellfish. More recently, the focus of concern has shifted to hepatitis A and other viral diseases, although salmonellid gastroenteritis is still a significant problem. Alarm over the health risks from consumption of raw shellfish has heightened in recent months. For example, a recent editorial in the New England Journal of Medicine urged that people cease eating raw shellfish altogether because the risks of gastroenteric illness are simply unacceptable (Du Pont 1986). This sentiment is shared by much of the medical community (Eckholm 1986).

Tomales Bay, located about 50 miles north of San Francisco, houses a small cultivated oyster fishery. The main threat to the fishery comes from effluent runoff from the surrounding dairy industry. Under normal conditions, runoff from the manure spread over each dairy's disposal area will have a negligible effect on water quality and hence on the fishery. During severe rainstorms, though, the manure will be washed into the watershed. The resulting fecal contamination of the bay will lead to contamination of the oyster beds. If the water quality around the oyster beds fails to meet public health standards, the fishery will be closed. Current standards, expressed in terms of the most probable number (MPN) of bacteria, are median values of 70 MPN/100 ml water for total coliform bacteria and 14 MPN/100 ml water for fecal coliform bacteria (Hunt 1977).

Contamination can be reduced by constructing holding ponds at the dairies. Each pond will retain effluent up to a fixed capacity. Because the manure floats, it will be washed into the watershed as soon as that capacity is exceeded. The severity and frequency of pollution will thus depend on the capacities of the holding ponds built. Current regulations require all dairies to maintain facilities to hold a 10-year 24-hour rainfall in addition a maximum average rainfall of 5.75 inches occurring in the previous three weeks (Rafter et al. 1974).

Hochman, Zilberman and Just (1977) analyzed optimal regulation of the Tomales Basin under the assumptions of zero acceptable risk and of uniformity of holding ponds capacities across dairies. In this case pollution is all-or-nothing: either

the bay is contaminated and the fishery shut down or the bay remains clean and the fishery stays open. Their analysis suggested that, to maximize the sum of fishery and dairy profits, dairies should be required to construct ponds capable of holding 50-year 24-hour rainfall events so that the fishery would be closed only one-fifth as often as under current regulations.

This characterization of the situation has several shortcomings. First, it ignores heterogeneity among dairies. Because of topoographical differences, the costs of constructing a holding pond of any given capacity tend to vary widely; Hochman, Zilberman and Just's data indicate that they may differ by a factor of four or more. This suggests that any given standard can be achieved at a lower overall cost by discriminating among dairies, i.e., requiring dairies in more favorable locations to build large holding ponds and those in less favorable locations to build smaller ones. Second, it examines only a simple, very stringent risk standard - essentially one of zero acceptable risk. This implies that the fishery must be closed whenever pollution is present. Thus, the analysis ignores the possibilities of substitution between closing the fishery and building holding ponds, which may permit standards to be achieved at lower cost. Moreover, it would be useful to examine the choice of a standard as well. For this purpose, it would be preferable to generate costs for a range of standards. Third, because the risk is all-or-nothing, uncertainty enters in a very restricted way, namely as the probability that a risk occurs. Examination of a broader range

of standards would have to incorporate much more of the probability distribution characterizing the risk. Fourth, only one source of uncertainty - rainfall - is modeled. There is also significant uncertainty about other factors affecting risk, including the amounts of dairy waste present, the limits between runoff and microbial contamination of the bay, the links between water quality and shellfish contamination and dose-response relations for illness resulting from the consumption of contaminated shellfish.

This study attempts to overcome some of these shortcomings. In what follows, we combine Hochman, Zilberman and Just's data on runoff control costs with a model of acute gastroenteritis risk from raw oyster consumption. We employ the approach described in the previous section to derive sets of uncertainty-adjusted, cost-efficient tradeoffs between total control cost and risk which we then use to analyze alternative regulatory approaches.

Risk of Acute Gastroenteritis from Microbial Shellfish Contamination

For simplicity, the risk of acute gastroenteritis from microbial shellfish contamination was modeled as a multiplicative combination of variables reflecting water quality, microbial uptake by oysters, and human physiological response. Two policy instruments were included: (1) a requirement that each dairy build a holding pond of a designated size, not necessarily identical, and (2) closure of the fishery.

Each cow was assumed to produce an identical constant amount of manure and each pound of manure was assumed to have a constant, identical microbial content. The microbial content of manure was assumed to be measured, accurately by the coliform count. Manure in runoff was assumed to be mixed uniformly in the bay waters around the oyster beds. This is undoubtedly an oversimplification, but does constitute a reasonable first approximation (see for example the data presented by the Northeast Technical Services Unit 1980).

Under these assumptions, water quality around the oyster beds will be proportional to the number of cows generating effluent. Runoff is affected by two things: (1) the sizes of holding ponds at the dairies, which is a function of policy, denoted X_1 and (2) rainfall, denoted R. Write the size distribution of holding ponds (that is, the number of dairies having holding ponds of capacity k) as $W(k, X_1)$ and the set of dairies generating runoff for any given 24-hour rainfall level r as $B[W(k,X_1),R]$. If there are J dairies having herds of sizes $H_1,\ldots,H_J$, then water quality can be written as $q\sum_{j\varepsilon B} H_j$, where q is a constant of proportionality.

To estimate q, it was assumed that during extremely heavy rainstorms effluent runs off at all dairies and that this effluent is the sole source of microbial contamination as measured by the fecal coliform count. This ignores the contributions from human sources such as defective septic tanks, which are important in certain areas but of secondary significance overall. The maximum fecal coliform count measured

over a broad range of oyster beds was 16,000 MPN/100 ml water (Northeast Technical Services Unit 1980). The total number of cows at dairies in the watershed, as reported by Hochman, Zilberman and Just was 13,200, implying a value of q = 1.2 MPN/100 ml water per cow.

Uptake of microbial contaminants by the oyster population was assumed to be proportional to water quality around the oyster beds. Letting u denote this uptake parameter, the proportion of oysters contaminated can be written as $uq \sum_{j \in B} H_j$. This parameter was estimated using data presented by Andrews et al. (1975) in their study of fecal coliform counts as an indicator of bacteriological contamination. They reported percentages of oyster samples testing positive for salmonella for five ranges of fecal coliform counts. These percentages were regressed on the midpoints of the fecal coliform count ranges without a constant term, with an additional assumption that the maximum observed fecal coliform count was 16,000 MPN/100 ml water, to obtain a estimate of u = 0.000057. The standard deviation of this estimate was 0.000022.

This approach has several shortcomings. First, it assumes that the presence of salmonella in oyster meat indicates a sufficient level of microbial contamination to cause illness. It would be preferable to measure the extent to which the meat is contaminated as well; however, the literature on dose-response relations looks only at whether or not the shellfish are contaminated, so this additional level of detail would add little of substance to the analysis. More significant is the implicit

assumption that the presence of salmonella is also a good indicator of viral contamination. The results of a number of studies indicate the contrary, specifically that the fecal coliform count in surrounding waters tends not to be such a good indicator of viral contamination (see for instance Gerba et al. 1980). However, no better indicator seems to be available at present.

A number of studies have shown that bacterial and viral populations remain relatively constant after processing and cold storage for several weeks (Kelley and Arcisz 1954; Wilson and McCleskey 1951; Erickson, Vasconcelos and Presnell 1967; Di Girolamo, Liston and Matches 1970; Hood, Baker and Singleton 1984). Because this study is concerned with incremental risk from water contamination, additional contamination from shucking and handling and improper storage were ignored.

The probability of contracting gastroenteritis from eating contaminated oysters was assumed to be proportional to the proportion of oysters contaminated. Letting d represent the dose-response parameter, this probability can be written as $duq \sum_{j \epsilon B} H_j$. Variations in this dose-response relation due to variations in the number of oysters eaten or the extent of contamination of each oyster were ignored. No data on bacteriological poisonings from consuming contaminated shellfish were known to us, so epidemiological studies of viral infections were used exclusively. Several such studies investigated incidents where guests at large parties had been served contaminated shellfish. The proportion of people consuming these

shellfish who contracted gastroenteritis ranged from .36 to .65 (Gill et al. 1983; Gunn et al. 1982; Morse et al. 1986). The dose-response parameter d was given a value .45, in accord with the incidence of gastroenteritis reported by Gunn et al. and the combined incidence of gastroenteritis and hepatitis reported by Morse et al.

Finally, the risk of gastroenteritis may be eradicated by closing the fishery and preventing harvest of the shellfish until they have decontaminated themselves. This effect can be represented by a function $o(X_2)$ which takes on a value of one when the fishery remains open and zero when the fishery is closed.

The risk of acute gastroenteritis from consumption of contaminated oysters, denoted t, can thus be written as $t = o(X_2)duq\sum_{j\varepsilon B} H_j$, a function of policies relating to the fishery, X_2, and to the construction of holding ponds, X_1.

As a first step, we will consider only one source of uncertainty, namely the randomness of 24-hour rainfall. Other sources of uncertainty will be incorporated in a future study. The probability distribution of 24-hour rainfall was modeled as a Pearson type II distribution as in Hochman, Zilberman and Just. Using data from the California Department of Water Resources (1974) they estimated that the 24-hour rainfall exceeded with probability b was 2.928 + K(b)0.413, where K(b) was chosen to correspond to a Pearson type III distribution with a skrewness of 1.3. Letting F(r) denote the probability that 24-hour rainfall R does not exceed a given level r, the risk associated with any given level of 24-hour rainfall can be written:

$$t = [1-F(r)]o(X_2)duq \sum_{j \varepsilon B[W(k,X_1),r]} H_j.$$

Calculation of Cost Curves for Gastroenteric Risk Reduction

Numerical methods were used to estimate total social costs and probability distributions of risk under alternative policies and to determine the minimum cost policy mix for each level of uncertainty-adjusted risk. Three levels of aversion to uncertainty were analyzed: neutrality, which implies regulation based on mean risk, and safety margins of 95% and 99%.

The Tomales Bay fishery produces a small share of total oyster production in the San Francisco Bay area; loss of the harvest was thus assumed to have a negligible impact on prices. The analysis performed by Hochman, Zilberman and Just suggested that holding pond construction costs would never be great enough to force dairies out of business, so this potential impact was ignored. It was assumed also that periodic closure of the fishery would have negligible long run impacts on production or employment. This characterization seems reasonable in light of the fact that most growers are small, individual operators using a minimum of equipment, so that entry is quite inexpensive. Finally, it was assumed that monitoring and enforcement costs are negligible, which also seems reasonable in light of the fact that the California Department of Health Services has only one person responsible for shellfish in all of Northern California. These assumptions imply that total expected social cost is closely approximated by the sum of amotized holding pond construction costs and expected losses in revenue from closure of the fishery.

The probability distribution of 24-hour rainfall events was partitioned into a number of discrete segments. For each of these states of nature the data collected by Hochman, Zilberman and Just were used to estimate the cost at each dairy of building a holding pond with capacity sufficient to hold no more than the corresponding 24-hour rainfall given the average maximum rainfall of 5.75 inches in the preceding three weeks. Following Hochman, Zilberman and Just, total construction costs were annualized by multiplying them by an interest rate of 10%. The result of these calculations was a set of functions for each dairy $C_1(b),\ldots,C_J(b)$ representing the annualized cost of building a holding pond with sufficient capacity for 24-hour rainfalls exceeded only 100b% of the time.

Expected revenue losses from closure of the fishery were estimated by multiplying the frequency of closures under alternative policies times the annual revenue of the fishery. Following Hochman, Zilberman and Just, fishery revenue was estimated at $500,000 per year. No adjustment for inflation was made in either case; thus, all costs are expressed in terms of 1973 dollars.

Costs and risks under a policy approach relying only on the construction of holding ponds under neutrality toward uncertainty were estimated as follows. Each dairy was assumed to behave as if it faced a tax T on effluent (effectively, on herd size). Constructin a holding pond with a capacity exceeded only with probability b will cost $C_j(b)$. The expected tax payments in this case will amount to bTH_j. Minimization of the sum of these two

costs implies that the optimal capacity for the j^{th} dairy, b^*_j, should be set to equate the marginal construction cost $C'_j(b^*_j)$ with the maximum tax liability TH_j. Construction cost for the j^{th} dairy is then $C_j(b^*_j)$ and total construction cost is found by summing up construction costs over all dairies. The choice of a capacity also gives a two-point distribution of effluent: effluent is zero with probability $1-b^*_j$ and proportional to H_j with probability b^*_j. Aggregation over the full set of dairies gives a probability distribution of total effluent which was then transformed into a probability distribution of risk using the model described in the previous section. Mean risk was calculated from that probability distribution, for each tax level, giving a risk, cost pair. Repeating this procedure over a range of values of T gave a set of such pairs, i.e. a cost curve for risk reduction.

For policies based on 95% or 99% margins of safety, the same procedure was followed with the addition of a constraint that holding pond capacity never be large enough for 24-hour rainfalls exceeded less frequently than the margin equal to P, this meant $b^*_j \leq 1-P$. The measure of risk used under these policies was the level exceeded only 100(1-P)% of the time, 5% and 1% for the 95% and 99% margins of safety, respectively.

The combined impact of the two policies of building ponds and closing the fishery was estimated using the same procedures with the additional constraint that pond capacity never exceed the frequency with which the fishery would be closed. Thus, if the closure policy were to prevent harvest after 24-hour

rainfalls no less than 100-yearevents, dairies were constrained to build ponds with capacities no greater than 100-year 24-hour rainfalls. If the fishery were closed with probability c, this implied a constraint of $b^*_j \leq c$ for neutrality with respect to uncertainty and $b^*_j \leq 1-P+c$ for a margin of safety P. The total cost of these policies was found by adding expected revenue losses to total construction costs as described above.

The efficient mix of policies and minimum-cost tradeoff curves for each level of aversion to uncertainty were estimated by comparing the total cost curves generated by these different closure rules. The resulting social cost curves for risk reductions were the lower envelopes of the total cost curves for each of these rules.

For policies under neutrality with respect to uncertainty, the cost of achieving a mean risk standard using uniform holding pond construction requirements was estimated by finding the frequency with which the maximum risk level would have to occur to generate the same average risk. Letting t* denote this maximum risk level, the capacity required to achieve any level of average risk t is $b^*u = t/t^* + c$, where c is the probability that the fishery is closed. Total construction costs for this uniform capacity bu* and revenue losses from closure were then calculated and compared as described above.

Such fine tuning is not possible for policies under 95% and 99% margins of safety: risk is either at a maximum or zero. The key problem in such cases is calculating the minimum cost combination of closure and construction required to achieve zero

risk. This was done by comparing the total cost curves for alternative closure policies using uniform building requirements.

Empirical Results

Some preliminary results of this analysis are shown in Figures 1-4.

Consider first the choice of policy instruments between closing the fishery and requiring construction of holding ponds. As Figure 1 shows, closing the fishery as seldom as 1% of the time is suboptimal except for very low risk standards. This occurs because closing the fishery is a relatively expensive way to reduce risk. It costs $5000 to reduce risk by 1% by closing the fishery. Building holding ponds becomes that costly only at very large capacities. For example, Hochman, Zilberman and Just found that, under a uniform building requirement, the marginal costs of these two instruments were equal only when holding ponds were capable of holding up to 50-year 24-hour rainfalls. The buildining requirements derived here are even less costly, implying that the marginal costs of the two instruments will be equated only at even larger capacities. Expressed in terms of risk, Figure 1 indicates that closing the fishery for 100-year and larger 24-hour rainfalls in cost-efficient under neutrality toward uncertainty only for risk standards of .001 and less.

The optimal policy mix changes as aversion to uncertainty increases. Figures 2 and 3 compare fishery closure with efficient building requirements under margins of safety for 95% and 99%, respectively. Closure is not used under a 95% safety

margin. This occurs because 24-hour rainfalls larger than 20-year events are of no concern to policy makers at this level of aversion to uncertainty. As we have seen, fishery closure becomes cost-competitive with building requirements only at much larger capacities. In contrast, closure is under for much higher risk standards under a 99% safety margin than under neutrality toward uncertainty. This occurs because very large, infrequent 24-hour rainfalls count very heavily under this level of aversion to uncertainty.

Now consider the impact of aversion to uncertainty on the cost curve for risk reduction. There are two exceptions to this. First, the 95% and 99% margin of safety cost curves are identical at maximum risk. This occurs because the construction of holding ponds has no effect on risk with these margins of safety until capacities of 20-year and 100-year 24-hour rainfalls are reached. Second, mean risk standards becomes more expensive than 95% margin of safety standard at very low risk levels. The reason for this is that decisions made under a 95% safety rule ignore risks that occur less frequently than 5% of the time, i.e., when very large rainstorms occur. Thus, a "zero" risk level is attained even though some residual risk actually occurs (albeit infrequently). It is precisely these residual risks which are the most expensive to eradicate. Under mean risk oriented regulation, their share of weight in decision making grows as the overall risk level declines, so that regulation becomes increasingly costly. These low-frequency risks do play an important part regulation with a 99% margin of safety, though,

with the result that the cost of mean risk standards never exceeds the cost of 99% margin of safety standard.

Excluding these two cases, it can be seen that the premium paid for reductions in uncertainty declines as the risk standard decreases. For example, achieving a 99% margin of safety costs about $10,000 more than achieving a 95% margin of safety for higher risk levels. This implies an average "uncertainty premium" of about $2500 for each 1% increase in the margin of safety. For very low levels of risk, though, this difference decreases. The gap between cost under neutrality toward uncertainty and under a 95% margin of safety is even greater for higher risk levels, about $80,000 for a risk of 0.35, for example. It decreases much more rapidly, though, and declines to about $5000 for very low risk levels. Neutrality toward uncertainty in this model corresponds to a safety margin of 45%, implying that the "uncertainty premium" averages about $1,600 for each 1% increase in the margin of safety for very high risk levels and falls as low as $100 for very low risk levels. This suggests that the cost of aversion to uncertainty increases as the level of aversion to uncertainty rises, a result consistent with the standard literature on risk and uncertainty.

Finally, consider the increase in social cost from the use of uniform holding pond size requirements such as are now in force. Preliminary calculations suggest that the gap between the minimum cost of risk reduction and cost under a uniform building policy is greatest at high risk for all levels of aversion to uncertainty and that the two converge as the risk standard

approaches zero. The latter result occurs because, to attain a zero risk standard, it is generally necessary to require holding ponds of the maximum relevant capacity, effect a uniform building requirement. Under neutrality with respect to uncertainty, the additional social cost of a uniform policy is about $55,000 at high risk levels, declines rather rapidly to $2,000 as risk declines and disappears as risk approaches zero. For safety margins of 95% and 99%, this additional cost reaches only about $7,500 and $5,000 respectively, suggesting that efficiency losses decline as aversion to uncertainty increases.

Concluding Remarks

We have argued that uncertainty tends to matter a great deal in environmental policy problems - including those concerning marine pollution - because of the incompleteness of the knowledge base from which decisions must be made and because of the expressed preferences of the general public. This implies in turn that formal methods for making environmental policy decisions should incorporate uncertainty explicitly. We have proposed once such methodology, which applies a safety rule decision mechanism to a probablistic model of environmental risk to generate uncertainty-adjusted cost curves for risk reduction. We argued that this approach combines several theoretically appealing features and practical usefulness.

The empirical analysis of microbial contamination of oysters in Tomales Bay by dairy wastes bears out these contentions. Calculation of the uncertainty-adjusted cost curves was feasible

and the curves behaved as expected: they were decreasing in risk and increasing in aversion to uncertainty. Aversion to uncertainty was shown to have a significant influence on the choice of policy instruments: closing the fishery turned out to be cost-efficient only for very high levels of aversion to uncertainty.

The empirical example also demonstrated some interesting characteristics of decisions made in this fashion. The premium in social cost required to achieve reductions in risk and increasing in aversion to uncertainty.

Regarding pollution control measures, the analysis suggested that inefficiencies due to the imposition to uniform control technologies tend to be greatest for high risk levels and low aversion to uncertainty. This result has some interesting implications for broader pollution control policies.

In sum, we believe that the analysis shows that our proposed methodology can be useful tool for environmental decision making and yields some interesting perspectives on a number of aspects of environmental policy questions.

Footnotes

[1]Although the information described in this article has been funded wholly or in part by the United States Environmental Protection Agency under assistance agreement CR811200-03 to the Western Consortium for the Health Professions, it has not been subjected to the Agency's required peer and administrative review and therefore does not necessarily reflect the views of the Agency and no official endorsement should be inferred. We would like to thank Dave Alton, Doug Price, Bob Hultquist and Mel Yee for their assistance. Responsibility for any errors is, of course, ours.

[2]This assumption is not very restrictive. Such representations can be derived for any probability distribution using Chebyshev's inequality or the partial moment methods proposed by Berck and Hihm (198?) and by Atwood.

References

Erik Lichtenberg and David Zilberman, "Efficient Regulation of Environmental Health Risks", Western Consortium for the Health Professions, San Francisco, CA, 1985.

Charincy Starr, Science.

Eithan Hochman, David Zilberman and Richard Just, "Internalization in a Stochastic Pollution Model", Water Resources Research 13:877-881, 1977.

Dale Morse, et al., "Widespread Outbreaks of Clam- and Oyster-Associated Gastroenteritis", New England Journal of Medicine 314:678-681, 1986.

O.N. Gill et al., "Epidemic of Gastroenteritis Caused by Oysters Contaminated with Small Round Structured Viruses", British Medical Journal 287:1532-1534, 1983.

Robert A. Gunn et al., "Norwalk Virus Gastroenteritis Following Raw Oyster Consumption", American Journal of Epidemiology 115:348-351, 1982.

R. Di Girolamo, J. Liston and J.R. Matches, "Survival of Virus in Chilled, Frozen and Processed Oysters", Applied Microbiology 20:58-63, 1970.

W.H. Andrews et al., "Comparative Validity of Members of the Total Coliform and Fecal Coliform Groups for Indicating the Presence of Salmonella in the Eastern Oyster, Crassostrea virginica", Journal of Milk and Food Technology 38:453-456, 1975.

M.A. Hood, R.M. Baker and F.L. Singleton, "Effect of Processing and Storing Oyster meats on Concentrations of Indicator Bacteria, Vibrios and Aeromonas hydrophila", Journal of Food Protection 47:598-601, 1984.

J. Erickson, G.J. Vasconcelos and M.W. Presnell, "Time-Temperature Effects on the Bacteriological Quality of Stored Shellfish", _____ 32:121-124, 1967.

Thomas E. Wilson, and C.S. McCleskey, "The Effect of Storage on Indicator Organisms in Shucked Oysters and a Comparison of Methods for the Determination of Enterococci", Food Research 16:377-382, 1951.

C.B. Kelly and William Arcisz, "Survival of Enteric Organisms in Shellfish", Public Health Reports 69:1205-1210, 1954.

Northeast Technical Services Unit, "Sanitary Survey of Shellfish Waters, Tomales Bay, California, February and March 1980", Davisville, RI, 1980.

Edward P. Larkin and Daniel A. Hunt, "Bivalue Mollusks: Control of Microbiological Contaminants", BioScience 32:193-197, 1982.

Herbert L. DuPont, "Consumption of Raw Shellfish - is the Risk Now Unacceptable?", New England Journal of Medicine 314:707-708, 1986.

Erik Eckholm, "Warnings Issued in Eating Shellfish that is Uncooked", New York Times, March 13, 1986.

R. Rafter et al., "Resource Conservation Data: Dairy Pollution Control Costs in Marin County, Minimum Guidelines for Protection of Water Quality from Animal Waste, Final Report", U.S. Soil Conservation Service, Marin County, CA, 1974.

California Department of Water Resources, "Summary of Short-Duration Precipitation Frequency in the San Francisco Bay Area", Sacramento, 1974.

C.P. Gerba, et al., "Bacterial Indicators and Environmental Factors as Related to Contamination of Oysters by Enteroviruses", Journal of Food Protection 43:99-101, 1979.

D.A. Hunt, "Indicators of Quality for Shellfish Waters" in A.W. Hoadley and B.J. Dutka (eds.), Bacterial Indicators/Health Hazards Associated with Water ASTM STO 635, 1977.

K.J. Thomson and P.B.R. Hazell, "Reliability of Using the Mean Absolute Deviation to Derive Efficient E, V Farm Plans", American Journal of Agricultural Economics 54:503-506, 1972.

E. Crouch and R. Wilson, "Regulation of Carcinogens", Risk Analysis 1:47-57, 1981.

Figure 1
Construction vs. Closure: Mean Risk

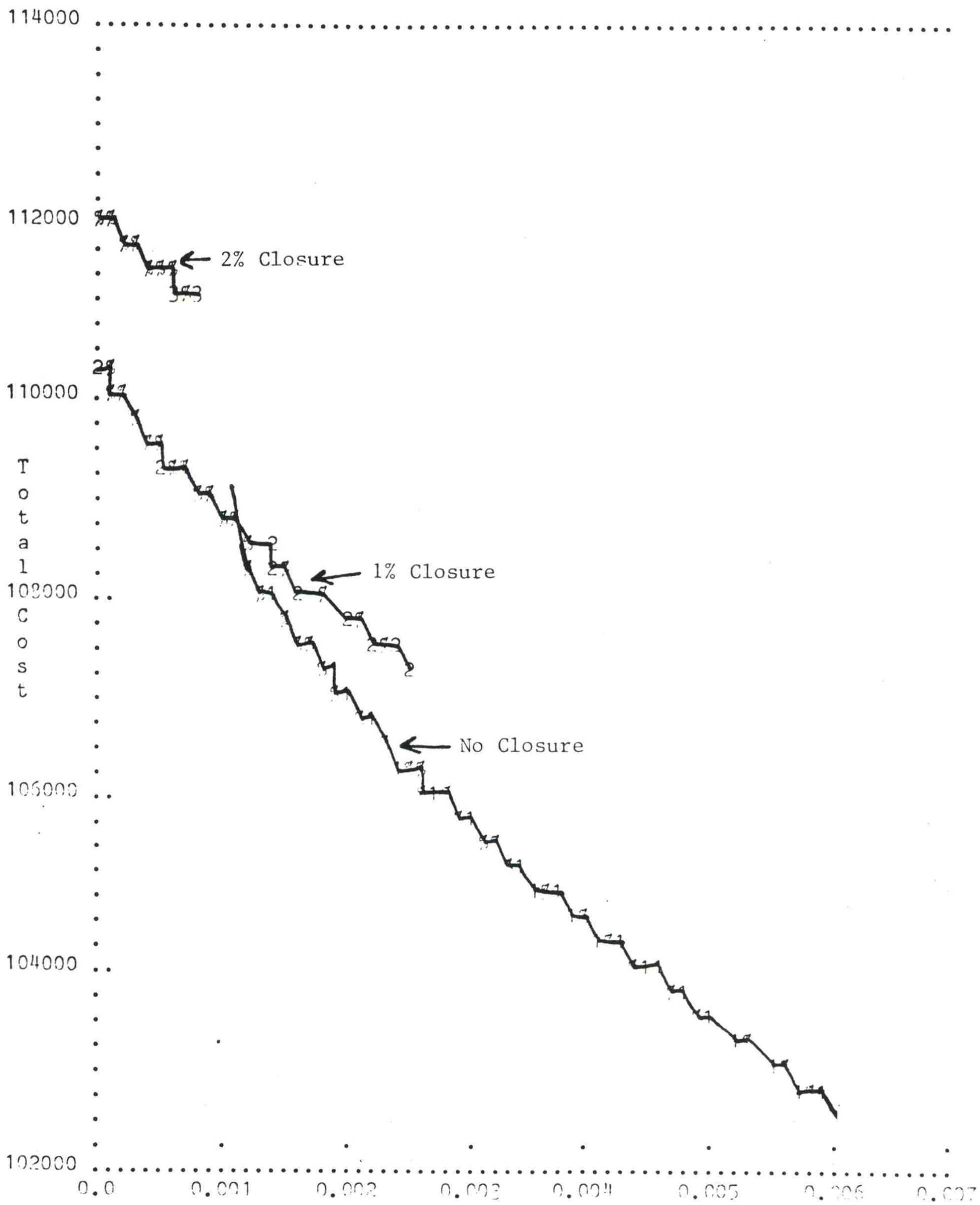


Figure 2

Construction vs. Closure: 95% Safety Margin

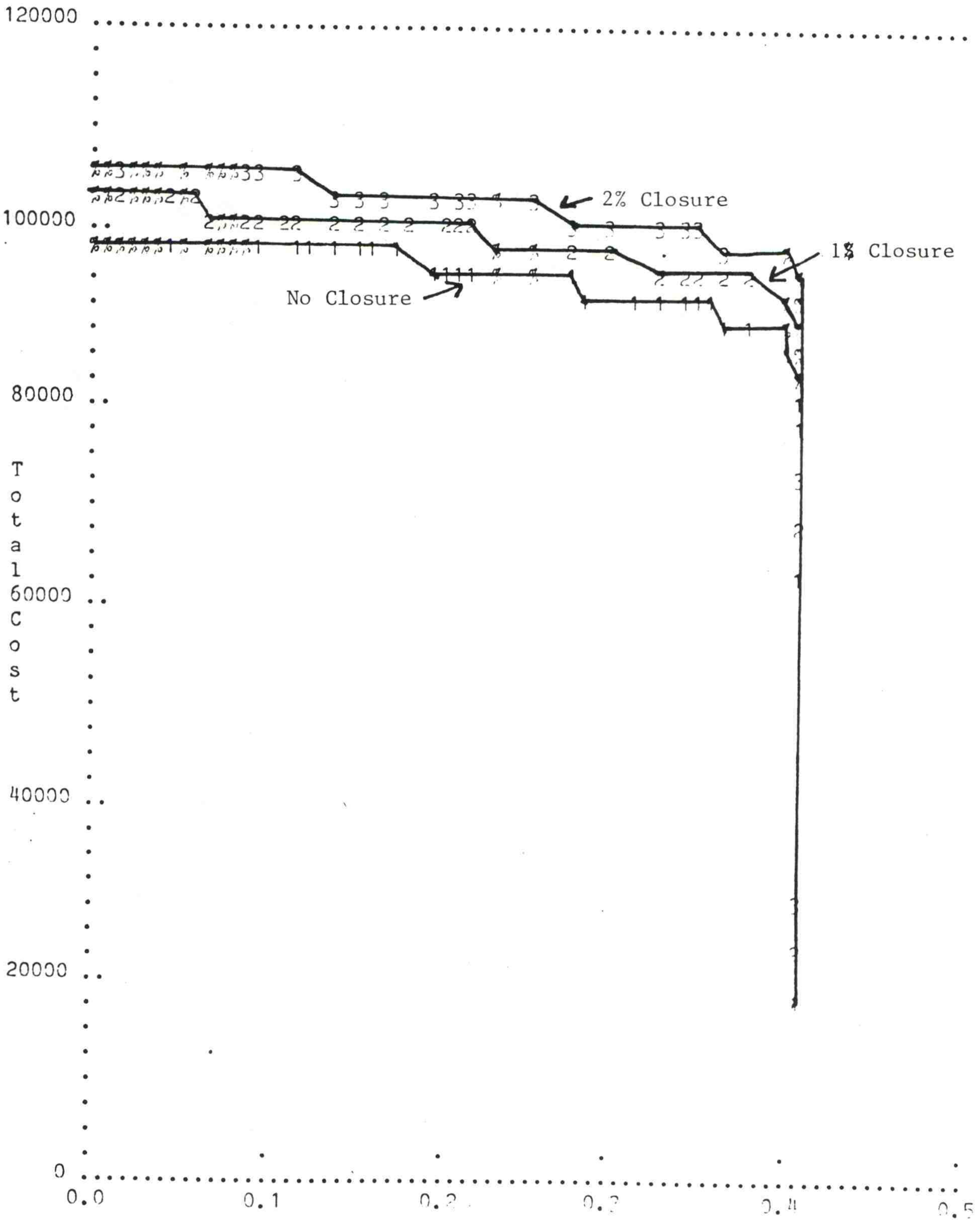


Figure 3

Construction vs. Closure: 99% Safety Margin

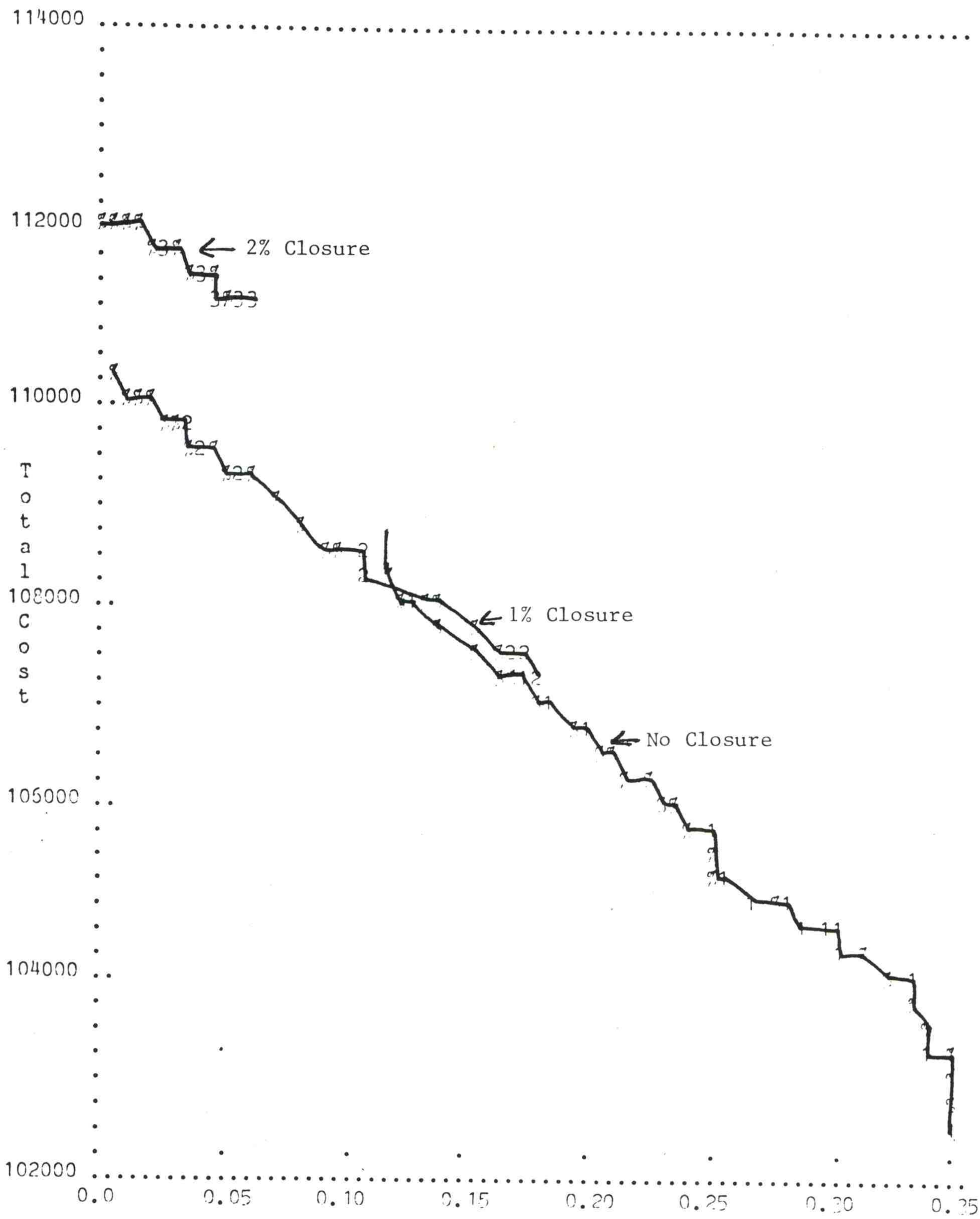
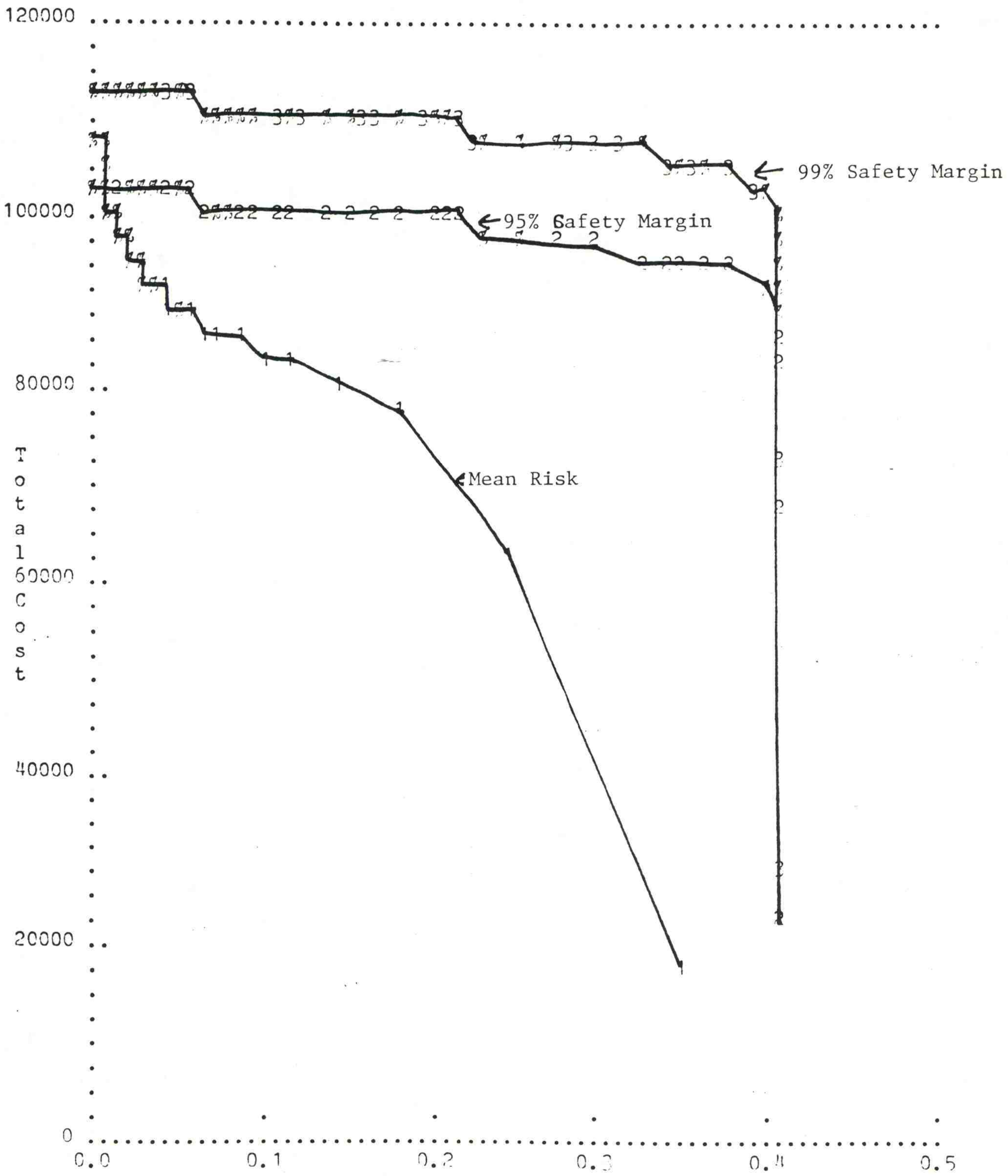


Figure 4

Impacts of Aversion to Uncertainty



WORKING DRAFT: DO NOT CITE OR QUOTE

Mitigating Damages from Coastal Wetlands Development: Policy, Economics and Financing*

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INTRODUCTION

In the last three decades it has become increasingly clear that the wetlands are an integral component of the ecosystem of the coastal marine environment. The ascribed role of wetlands in the marine environment extends beyond such obvious functions as nursery and habitat to the contribution of wetlands to the nutrient and energy budgets of estuarine systems. Nevertheless, the scientific evidence on the relationship of wetlands to fish and wildlife populations, to shoreline erosion control, and to water quality is still being developed and remains somewhat contradictory [Nixon, others]. Furthermore, at this time, the ability to identify the contribution of specific wetlands parcels to the functioning of the marine ecosystem is limited [Oviatt and Nixon, others]. However, the general importance of wetlands systems to the natural productivity of the marine environment is seldom a matter of scientific debate.

During the same period that the scientific evidence on the ecosystem role of wetlands was accumulating, it was also evident that the nation's wetlands stock had been depleted by direct development on former wetlands (e.g. San Francisco Bay) and by indirect degradation from economic development activities [OTA]. A particularly useful illustration of indirect degradation is provided by the current condition of the Louisiana coastal marsh. For thousands of years the Mississippi River has meandered over the landscape creating a pattern of bank erosion and streambed deposition over its entire length. As a result the river has carried a heavy sediment load to its mouth along the Louisiana coast. As the flow rate slowed near the River's mouth, sediments were deposited and a delta was formed which developed into extensive marsh lands. The pattern of sediment delivery reached a dynamic equilibrium with forces of subsidence and shoreline erosion; over time vast acreages of marsh were maintained. At the same time a mix of vegetation types emerged and an extensive pattern of wetlands environments, ranging from saline to freshwater, developed.

Beginning in the last century, economic development began to intrude on this hydrologic process. Upstream tributary reservoirs were constructed and served as sediment traps; bank erosion was controlled. The result was a reduction in sediment delivery [NWS]. At the mouth of the river, navigation channel development constricted the meandering channel and funneled sediment and

freshwater far offshore. At the same time, channels were cut through the marsh for pipelines to carry offshore gas and oil to on-shore processing facilities. These pipeline cuts affected the salinity regimes. Today as a result of development induced sediment starvation and changes in the salinity regime, Louisiana wetlands are disappearing at a rate far in excess of what would otherwise be the case. Subsidence and erosion are no longer being checked by sediment deposition and vegetation losses, and, as marsh changes from fresh to saline with salt water intrusion marsh soils are left unanchored by vegetation and subject to erosion [Watson, 1984].

Until recently state and federal policy encouraged "reclamation" of wetlands for commercial use. However, during the last three decades, in response to the changing views on the importance of wetlands, this policy position has been modified and, at all levels of government, programs have been adopted to manage the rate and location of wetlands alterations. However, because scientific study of the relationship of wetlands to the marine environment is a relatively new endeavor, the establishment of scientifically defensible wetlands management goals and strategies of wetlands management has been difficult. In addition, there have been cases where damages to coastal marsh have occurred and legal actions have been taken to secure compensation for such damages. In those instances where wetlands damage has occurred, the basis for determining how much, if any, compensation is required also has been difficult.

We will first describe the technical obstacles to measuring the economic value of a coastal marsh environment. The focus will be upon the absence of scientific understanding and data suitable for sound valuation efforts. Next we will describe the rapid evolution of the nation's wetlands management programs. We will argue that the resulting policy has developed without a clear statement of goals and without a cost effective management program. One effect of this condition is that there is an ambiguous basis for ascribing economic penalties for damages to coastal marsh environments.

With this as background, we will propose a framework for wetlands management and damage assessment. Specifically, a policy based upon setting targets for maintaining regional wetlands stocks will be described, and the establishment of a wetlands development fee and wetlands banks will be outlined. The problem of wetlands valuation will be addressed in this management context, and a defense will be offered for using the cost of wetlands construction and rehabilitation as a basis

for "value" estimation. The last section of the paper will illustrate the application of the management and damage assessment approach for the Louisiana coastal marsh.

Valuing and Managing the Undeveloped Coastal Marsh: The Standard Efficiency Framework

The valuation of a natural wetlands can not be made by inference from land market prices because the value of many of the service flows will not be appropriable by the land owner. Thus valuing a marsh parcel as a fixed asset, which will yield a flow of services, requires valuing the service vector of the marsh parcel in question and then inputing the values of the service to the marsh itself. The general framework for evaluation of natural wetland areas is discussed below in the context of Figure 1.

The framework for valuation of unaltered natural wetlands builds upon the recognition that a wetlands area can be viewed as a physical asset which functions as part of both a hydrologic and ecologic system. In turn this functioning gives rise to at least some of the following services: recreation, water supply, erosion control, and wildlife and fish habitat. The possible existence of these services means that an economic use-value for a wetlands area may exist if the one or more of its services is scarce. Figure 1 depicts a valuation sequence from wetlands area (Box I) to wetlands function (Box II), to wetlands service (Box III), to service use-value (Box IV) along linkages (a), (b) and (c). Wetlands values in Box IV are money equivalent measures of value in terms of economic surplus. Wetlands valuation requires a basis for quantifying each of these linkages.

Also depicted in Figure 1 is the possibility that substitutes may exist for a wetlands service. Substitutes for a wetlands area (Box V) include construction of new wetlands by whatever means are technically available to replace a wetlands lost to development. The constructed wetlands would be presumed to provide similar functions as the lost area. Purchase and presentation of wetlands *which would otherwise be lost to development* may also be considered a substitute for a natural wetlands area. Substitutes for wetlands functions (Box VI) would include, but not be limited to water control structures, artificial ground water recharge, erosion control structures, water-water treatment plant and fish hatcheries. These are possible substitutes which can give rise to a hydrologic or ecologic function that might equally be derived from the wetlands area. Substitutes

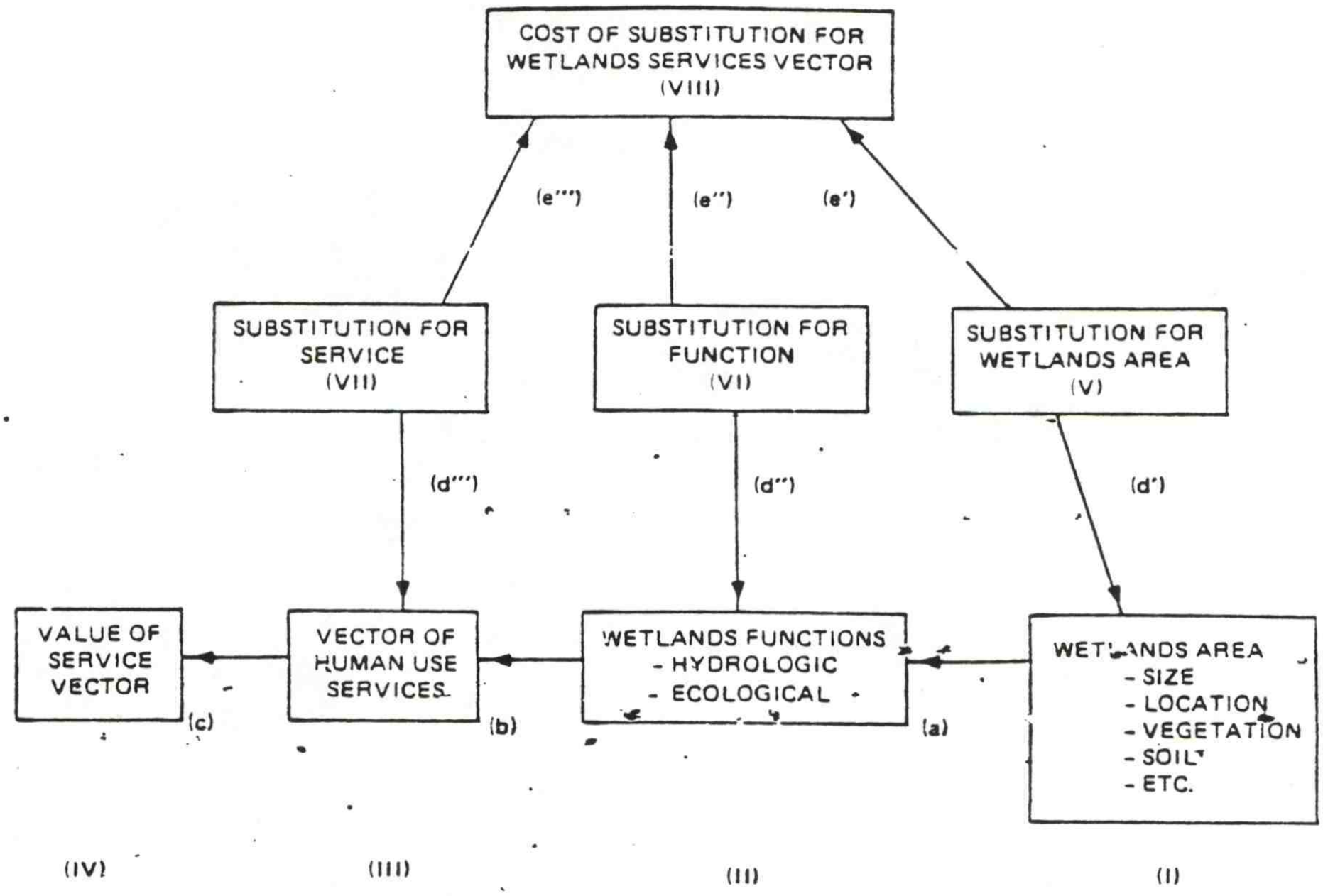

Figure 1. Determination of unaltered wetlands value.

for wetlands services (Box VIII) include all those possible actions that directly provide a service that can be provided by wetlands function. Such service substitutes would include but not be limited to, changes in location of economic activity to less flood prone or erosion prone land, provision of water from alternative sources or water conservation, harvest of commercial products from alternative areas and alternative recreational activities and sites.

The linkages (a), (b), (c), (d′), (d″), and (d‴) suggest that the wetlands services available at any time and place will depend upon the mix of wetlands areas with area, as well as function and service substitutes. Linkages (e′), (e″) and (e‴) suggest that the cost of wetlands substitution (Box VIII) depends upon the particular mix of actions taken from Boxes V, VI, and VII.

The economic approach to valuation of a natural wetlands area can be readily shown with Figure 1. The basis for establishment of wetlands area value is the ″with and without″ principle. Specifically, to the extent that a natural wetlands area is the only source of wetlands services, the foregone economic value of these services (Box IV) with the change in wetlands area, as compared to without the change, can be entirely attributed to the wetlands area change along arrows (c), (b) and (a).

Of particular importance is that the ″with versus without″ principle of valuation requires that valuation be specific to each area where wetlands′ change is being considered. Site-specific analysis is implied for at least two reasons. First, a wetlands may provide a service but no use-value will be lost if the wetlands area is developed. For example, a wetlands that is recharging groundwater might have no water supply use-value if all water users in the area are dependent upon an upstream reservoir for water supply. Second, not all areas are equally capable of providing a wetlands service. Thus to observe that a service exists with the wetlands area does not logically permit the analyst to conclude that the service will cease to exist without the wetlands area. For example, a particular commercial fish harvest level which exists with a wetlands area may not change when the wetlands are developed, if changes in fish populations with the wetlands alteration are too small to affect harvest. Continuation of small reductions in wetlands acreage over time may result in loss of a service if the cumulative losses reduce wetlands acreage to below some threshold level needed to maintain the service level. At this point positive marginal values for each remaining acre will rise.

The "with versus without" argument can be described as the isolation of the value of a marginal acre of wetlands. Documentation that wetlands on average may provide a wide array of services does not serve to establish marginal values for specific land parcels. Furthermore the "with and without" principle requires that the measure of natural wetlands value must consider whether wetlands substitutes exist. If substitutes do exist, then the cost of employing substitutes to replace the wetlands service must be considered as an alternative measure of value. This is traced along arrows (d′), (d″), (d‴) and (e′), (e″), (e‴). The value of a wetlands area will be the lesser of (1) the least cost combination of wetlands substitutes capable of providing the same services, (Box VIII) or (2) the direct measure of service value attributable to the area (Box IV). This is so because people would not be willing to pay for a wetlands any more than the lesser of the value of the services it provides or the cost of replacing the services by employing a wetlands substitute.

It is, therefore, also the case that the cost of replacing the services is the upper limit on wetlands value. The standard criticism of using replacement cost is that it can easily overstate economic surplus measures, unless there is evidence that people would be willing to pay for the wetlands service if it were sold (presumably by a discriminating monopolist) at prices sufficient to cover the cost of the substitute. The fact that replacement cost is an upper bound on economic value measurement is not as commonly acknowledged.

When identifying the cost of a wetlands substitute, consideration should be given to the "least-cost" substitute. In finding the least-cost alternative combinations of service, function and area substitutes, combinations would be chosen and implemented in stages so that the wetlands service vector is replaced at lowest cost. This search for the the least-cost alternative is necessary since the presumption is that people would only be willing to pay an amount equal to the least-cost way of replacing the service if it were lost. However, identification of the least-cost alternative combination will require knowing that service, function and area substitutes could substitute for the wetland area being evaluated. In fact, the wetlands area, now or at some future date must be likely to provide the service flow. For example, purification of wastewater can only be a value for a wetlands if the area in fact receives and processes wastewater. If an area does not receive wastewater, then it would need to be demonstrated that ambient water quality in an area would decline without the specific wetlands parcel.

As an alternative to identifying the least cost combination of substitutes, the physical substitution of a wetlands area can be presumed to replace whatever services were flowing from that area without having actual knowledge of linkages in Figure 1. The wetlands assessment process needs to determine the structural features of the wetlands area to be valued as a basis for insuring substitution of ecological and hydrological function. Then, it can be presumed that the service vector of the substitute wetlands will be identical to the service vector of the area being valued. However, the service vector itself (Box II, in Figure 1) need not be known. Cost of a wetlands area substitute can be estimated by standard engineering cost estimation methods. These cost estimates would be a *maximum* measure of the value of the service vector (Box IV) and, hence, the wetlands area being replaced (Box I). This represents a maximum value measure because it will be equal to or greater than the value of the service vector as well as equal to or greater than the least cost combination of wetland substitutes.

It should be clear that accurate use valuation depends upon extensive wetlands function analysis. First, if natural wetlands services value is to be imputed to a wetlands area, linkages (a), (b), and (c) in Figure 1, it is necessary to identify services produced by a specific area and the level of service provision. Otherwise, the analyst may establish a potential natural wetlands services value (Box IV), for example for recreational fishing, but not be certain that the wetlands area being valued is providing that service. A second reason for needing function and service assessment analysis is to be able to specify whether substitutes are able to produce the same services as the wetlands area being valued -- linkages (e′) (e″) (e‴).

Within this context, what are the prospects for establishing the asset value of a natural wetlands acre? Practical limitations on the application of the valuation methods available for natural wetlands services are numerous. Such valuation requires detailed function and service assessment studies for specific wetlands areas in order to establish whether such areas are capable of producing a particular wetlands service. Since all wetlands are not of equal productivity, the needed detailed assessment analysis will be both time consuming and costly. These assessment needs will stretch the ability of wetlands scientists to make scientifically defensible claims about the services of any given wetlands in the larger marine environment.

Oviatt *et al.* [ ] after reviewing the knowledge of ecosystems functions and services of coastal marsh in Rhode Island offer an assessment of the difficulties posed a measurement goal. They state:

> We conclude that the evaluation of individual coastal salt marshes in terms of their environmental contribution is not likely to lead to successful management strategy. (p. 210)

They reach this conclusion from a perspective which argues that individual wetlands areas do not exist in isolation from each other in terms of the hydrologic or ecological functioning. There is a synergistic effect among marsh areas as opposed to an additive-effect and, therefore, they conclude that "development of ecological rating systems for coastal wetlands must be viewed with considerable skepticism, for there is little reason to believe that such systems can be established on sound scientific ground." (p. 211)

When it is recognized the economic evaluation of a marsh assets will depend upon the scientific base of the ecosystem argument it is essential to recognize the concluding comments of Oviatt *et al.* on the scientific foundation for wetlands management.

> It [wetlands management] should not be made on the basis of an elaborate rating scheme that is, in fact, built on a very shaky intellectual foundation. It is a great disservice to pretend to so much certainty when we are still far from knowing what is happening in the wetlands. (p. 211).

Application of economic valuation techniques in individual cases will require substantial analytical cost and time requirements, especially as more complex theoretical considerations must be made. These cost and time requirements mean that for numerous small wetlands areas, careful valuation of services will be infeasible. Unfortunately, general value estimates are not applicable to individual areas because of the site specific differences in productivity of wetlands areas and the site specific nature of the demand for wetlands services.

Despite these measurement difficulties, a perception persists among economists and others, that it is necessary to ascribe economic values to individual wetlands areas. Such a perception persists within the economics discipline because of the strong orientation to the economic efficiency perspective on resource allocation which requires analysis of marginal gains and costs over a continuing sequence of decisions. However, there are other opportunities for economic analysts to

serve wetlands policy and management. The scope of this alternative economic work is discussed in detail after a brief review of current wetlands programs.

The Wetlands Policy Context

The nation has no explicit policy governing the protection of wetlands from development. Rather wetlands policies and programs are the product of a diffuse and evolutionary process that implies favoring wetlands protection over wetlands alterations. The rate of wetlands alteration has slowed substantially during the last two decades, but the losses each year remain substantial [OTA]. In part these losses are the result of a continuing effect from previous wetland destruction (such as that affecting the Louisiana coast), where there is not an explicit program for restoration. At the same time, because regulation proceeds on a case by case basis, wetlands regulation has been only a partially effective tool in a diffused and contradictory regulatory environment. In a recent speech, Milton Russell, Assistant Administrator for Policy, Planning and Evaluation at the EPA, summarized the situation by noting that "our current wetland policy is confused and often inadvertently destructive" (p. 8). Such a conclusion is consistent with the findings of other studies [OTA].

It is instructive to briefly review how the national policy developed and to note its fundamental differences with other environmental programs. In the late 1960's a series of court interpretations of the 1899 Rivers and Harbors Act required that the U.S. Army Corps of Engineers expand their review of dredge and fill permit applications to include not only obstructions to navigation, but also the effects of fill activities on wildlife habitat. This judicial action was intended to bring the Corps' permitting program into compliance with the requirements of the Fish and Wildlife Coordination Act of 1958 (FWCA). However, the FWCA required only that of habitat effects be considered in decisionmaking; there was not a no mandate to protect habitat.

Questions which arose immediately were whether the limited jurisdiction of the Corps' permit program, to the navigable waters of the United States, included wetlands adjacent to all water bodies. Another matter in need of interpretation was whether the effects on habitat were to be only at the site or were to also include possible indirect effects of filling activity. The passage of the National Environmental Policy Act (NEPA) of 1969 served to expand the required review of permits to environmental concerns beyond wildlife habitat, if such a permit was deemed to be a

"significant" federal action. However, NEPA, like the FWCA, only required that consideration be given to environmental impacts and carried no substantive statement of environmental goals.

Although legislative action to clarify the national policy to wetlands development would have been desirable, the actions of the Congress in the 1972 amendments to the Federal Water Pollution Control Act (FWPCA) did not clarify wetlands policy; yet it is Section 404 of the Act which is the basis for the existing federal wetlands program. In Section 404 the Congress expanded the Corps' permit authority to require that permits be denied when permitted activities would adversely affect either navigation (under the 1899 Act) or water quality standards as established in compliance with the FWPCA of 1972. (There were some exceptions to the scope of this regulatory authority). Proponents of wetlands protection subsequently filed a series of court cases to argue that there was a demonstrable link between wetlands and adjacent water quality and that, therefore, it was the intent of the Congress in framing Section 404 that the Corps be responsible for review of proposed development in all wetlands. At the same time the NEPA process and the FWCA requirements remained in effect, and the conclusion often was made that an overall federal wetlands protection strategy had been pieced together.

However, there was no concurrence among the federal agencies, among the states, or within the larger public that Section 404 was intended as a wetlands protection program. For example, the Congress left unaddressed issues of jurisdiction-- e.g. which wetlands are part of the navigable waters and whether indirect effects of filling were to be considered in permit review. To this date there remains a debate over whether it is only the direct water quality effects of wetlands filling which are covered by Section 404.

Wetlands protection efforts have been analogous to the manner in which wilderness and national park policy is made without any definition of standards to be achieved; in that regard it was unlike water quality management which is directed by stream and effluent standards. Although this discussion has focused upon the federal program, the problems in the states were analogous ones and legal indeterminancy and scientific uncertainty have resulted in a variety of different approaches to wetlands management. The result of the national process leading to the current wetlands policy setting is that most attention has been paid to regulatory tactics both by those who wish to restrict wetlands alternations and by those who feel some wetlands conversions should be

accommodated. A discussion of the twists and turns of this process, usually in the courts, would be interesting but not germane. What has been missing is a clear statement of goals and standards for wetlands regulation to give a larger context to the regulatory framework.

Regulatory critics have argued (with justification) that, at times, Section 404 regulation has been unresponsive to cost considerations [RIA] and often inflexible even when those who wish to develop wetlands offered compensation [Tenneco]. Without a more complete statement of the purposes of the wetlands regulation it is unlikely that these objections to the program will be overcome.

In part the regulatory dilemma has been a product of the fundamentally weak scientific understanding of wetlands relationship to water quality and other environmental services. Denial of a wetlands development proposal requires the permit agency to demonstrate the negative affect such development would have on water quality. If the agency wished to consider other possible environmental values, there was an equally weak scientific basis for linking a particular wetlands permit to the larger environment. As Milton Russell noted in his recent talk, "It would be easier if we knew more. If the various wetlands ecologies were really understood, we could make intelligent defensible resource decisions." (p. 9)

Directly analogous to the regulatory management problem has been the problem of assessing damages to wetlands. As awareness of the importance of natural environments has increased, there has been a number of court cases seeking compensation for damage to natural environments, including coastal wetlands. In one case, the defendants were the owners of the *S S Zoe Colocotroni*, a tramp oil tanker which ran aground off Puerto Rico in 1973. In order to refloat the ship, the captain jettisoned over 1.5 million gallons of crude oil. The oil came ashore in the Puerto Rican bay of Bahia Sucia where, despite cleanup efforts, there was substantial damage to benthic and intertidal organisms and mangroves. The Commonwealth of Puerto Rico and the Environmental Quality Board sued for damages (Commonwealth, 1980). The court, after hearing various competing views on the calculation of damages, awarded the plaintiffs $6.2 million. This damage was determined by estimating that there had been a decline of 4.6 million organisms per acre due to the oil spill; then, a price of 6 cents per organism charged by biological supply laboratories was used to compute an organism replacement cost of $5.5 million. The judge then

added to this figure the costs of replanting 23 acres of mangroves and the cost of oil cleanup to derive the $6.2 million total. This damage award was appealed and the higher court struck down the computation of damages by use of prices from biological supply catalogs. The higher court concluded with respect to estimating damages that,

> To say the law on this question is unsettled is vastly to understate the situation... we....have ventured far into uncharted waters.. (and cannot)... anticipate where the journey will take us [Commonwealth, 1980, p. 46].

The case was remanded to the original court with the suggestion that the plaintiffs may wish to consider such items as alternative site restoration in computing money damages.

The *S. S. Zoe Colocotroni* case used the principle of replacement cost but the application was found inappropriate. In this case, the damages were measured by the cost of replacing one service (organisms) rather than the physical system that supported the organisms. It is extremely doubtful that society would demand the organisms, if available at $5.5 million, or that such a replacement was a least cost alternative. Indeed the higher court recognized these problems when they stated that awarding actual wetlands restoration costs is

> a far different matter from permitting the state to recover money damages for the loss of small, commercially valueless creatures which assertly would perish if returned to oil-soaked sands, yet probably would replenish themselves naturally if and when restoration - either artificial or natural - took place [Commonwealth, 1980].

Recently, another case focused, in part, on wetlands damage. In 1979, Louisiana placed a "first use tax" on natural gas which was pumped off-shore but which was "first" processed in Louisiana before being transported to out-of-state markets [Louisiana, 1979]. Louisiana claimed that the natural gas processing and transportation facilities had caused severe damage to the state's coastal wetlands and that the tax proceeds were designated to restore and maintain the wetlands.

It was on this basis that Louisiana claimed that tax collections of $264 million per year were "fairly related" to wetlands damages. The states of Maryland, Illinois, Indiana, Massachusetts, Michigan, New York, Rhode Island, and Wisconsin sued to stop the tax from being implemented. As one argument in the case, the plaintiffs claimed the tax levy exceeded environmental damages.

While it was relatively easy to establish that some wetlands damage had occurred--particularly due to pipeline placement and right-of-way maintenance--the question of assigning damages for

establishing the tax was an important basis for the court challenge. On June 15, 1981, the U.S. Supreme Court struck down the Louisiana "first use" tax as unconstitutional and ordered all previously collected taxes refunded with interest; however, the decision was not based upon a consideration of whether the tax was fairly related to environmental damages. Therefore, the basic question of assigning damages remains an open one for future cases of this type.

Toward a Rationalized Wetlands Policy

Given the current state of wetlands science and national wetlands policy, what contribution can economic analysis make to an improvement? We have argued elsewhere that the current state of understanding of wetlands asset values, as distinct from individual service values, makes a case for a policy bias favoring wetlands preservation unless the costs of such a bias are unacceptably large in any specific instance [Batie and Mabbs-Zeno, Shabman, Batie and Mabbs-Zeno, Shabman and Bertelson]. We also would argue that a policy based upon a benefit-cost balancing test for wetlands permitting is technically impractical. Finally there is sufficient evidence that, despite the ambiguity of many aspects of wetlands policy, there has been a national shift from viewing wetlands as wastelands to viewing wetlands as national assets in need of protection. The shift in viewpoint is a reflection of fundamental realignments of the recognized implicit and explicit property claims to use of these landscapes. These realignments are away from solely private discretion in determining the fate of a land parcel to a sharing of that decision with regulatory authorities. The essence of these general arguments is that the economic policy question of interest is not whether wetlands should be preserved but rather over what the most efficient manner to achieve that goal. Milton Russell summarized his views on this argument by concluding

> responsible position, it seems to me, is to avoid casual, uncaring destruction. It is to raise the hurdle' over which those who want to convert or otherwise damage wetlands must jump. In time research may show that these hurdles can be lowered. But unless we set them high now, wetlands research will be of merely academic interest... (p. 10)

To frame an economic policy problem in this context requires that a wetlands goal be established and that institutional reforms be designed which are able to achieve that goal at least cost. This same perspective underlies applied environmental economists' arguments in support of effluent taxes and transferable pollution rights (TPR). There is a potential to transfer the logic of those proposals from their typical application to water and air quality to wetlands management.

In order to make the rationale for this transfer more clear, it is necessary to briefly reflect upon how, for example, a effluent tax system is supposed to work.

In the effluent system the policy decision is divided into two separate parts (although feedback over time can cause the two parts to become interdependent). In the first part, an ambient environmental goal is established for a particular area such as a river segment. The ambient goal is in turn translated into a maximum allowable waste discharge to the area the will cause realized environmental quality to remain within the stated goal. While economic benefit analysis may be a part of the process which defines the goal, the benefit analysis is not expected to be a definitive guide.

With an ambient goal established, the second aspect of the policy is to design an institution to allocate waste reduction requirements such that the marginal cost of waste withholding is equated across all waste dischargers. In the effluent tax system, the price is set administratively to achieve this marginal cost pricing rule. At the same time, the effluent tax reflects the marginal cost of waste withholding for the last firm which chooses to pay the tax.

For purposes of this argument, it is worth considering the economic implications of this approach in more detail. Specifically, the effluent tax is equal to the marginal cost each party bears to maintain the ambient standard. Stated differently, the effluent tax is equal to the marginal cost of compensating for an increase in discharge at one point by a reduction at another in order to maintain the ambient standard; the effluent tax is equal to the cost of one party *replacing* anothers waste withholding effort. As long as the cost for treating the waste rather than forgoing output for changing production practices is the waste withholding cost, then a measurement of the marginal cost of waste treatment is a sound basis for setting the initial effluent fee. In any event, this is the recommended approach for setting an initial fee [Kneese and Bower]. In short, replacement cost is the accepted basis for a practical effluent tax scheme.

There are a number of practical reasons to build a wetlands policy on the replacement cost basis. In addition such an approach is analogous to the typical effluent tax proposals favored in applied environmental economics studies. In establishing a wetlands policy on a replacement cost foundation it must be established that wetlands are replaceable either by restoration efforts or by

the creation of new wetlands. Indeed, the evidence is accumulating that wetlands construction and replacement is a technically achieveable practice that should be integrated into any wetlands management program. The feasibility of wetlands construction and rehabilitation has been a subject of recent research and practical experimentation [Dunston et al. 1975, Saucier 1978, Garbish 1978, Cole 1979]. Preliminary investigation of marsh creation in coastal areas suggest that constructed march areas offer many of the same services as natural marsh after one to three years [Garbish 1978, Jerome 1979, Newling 1981]. Ashe [1982] argues that restoration of previously altered areas may be a more feasible and suitable means of maintaining a wetlands base than new wetlands construction. Areas physically altered in the past are easily identified and may only require minor amounts of remedial engineering activity to restore wetlands functions and services.

The physical replacement or restoration of one wetlands area can be presumed to replace whatever services were flowing from another area which was developed, without having actual knowledge of linkage (a) in Figure 1. The biological assessment process need only determine the structural and hydrologic features of the wetlands area to be replaced as a basis for insuring physical replacement of those features. Then, it can be presumed that the service vector of the replacement wetlands (along arrow d) will be identical to the service vector of the replaced area (along arrow a). However, the service vector itself (Box II, in Figure 1) need not be known. Costs of wetlands replacement or restoration can be estimated from standard engineering cost estimation methods.

Wetlands management must be targeted to an eco-region [FWS] much as water quality management can only be pursued in a specific watershed context. However, the possibility of demonstrating conclusively the relationship of individual wetlands parcels to the larger eco-region is limited at best. As a result, wetland policy reform must begin by initiating a process of goal setting for maintenance of minimum wetlands stocks of various types [FWS] within defined eco-regions. In principle all wetlands may be replaceable, however, certain areas would no doubt be reserved from development (e.g. "wetlands wilderness areas") in the setting of a wetlands policy. Given the historical loss rate of wetlands, and the continuing scientific uncertainty about their relationship to the larger ecosystem, a bias toward preservation of something close to the present wetlands stocks seems warranted. The role of economic benefit analysis in such a process is not

clear, however, basic opportunity cost and marginal value principles can help structure the discussion and debate over goal setting. To quote Milton Russell once more,

> there is not enough known to do that [set a wetland stock goal], it strikes me that the only sensible policy is to start pushing harder on the brakes, wherever wetlands are threatened. (p. 10)

This recommendation to begin by goal setting is made neither casually or naively. The costs of developing the information base needed to inventory wetlands and mapping their location will be substantial. However, such an effort has been underway for a number of years within the Fish and Wildlife Service. In addition the maintenance goal will not be expected to be static over time, but rather would respond to new scientific information on wetlands and to recognition of the actual replacement costs of maintaining a goal (actual replacement costs are discussed in detail below). Such a dynamic approach to goal setting is of course descriptive of the type of trial and error process that always characterizes decision making where neither technical or value information is ever attainable [Wildavsky].

Wherever wetlands can be created and restored it is possible to permit actions which destroy wetlands as long as offsetting actions are taken to reestablish a wetlands at another location in order to maintain the wetlands stock. This is the exact logic behind using effluent fees and TPR systems to maintain an ambient environmental standard. The replacement action may be taken by the wetlands developer as part of the regulatory agreement which yields the permit. Replacement may be by purchase and preservation of wetlands which would otherwise be lost to development or by construction/rehabilitation of another wetlands areas. In fact, such mitigation requirements are occasionally made a condition of wetlands conversion permits. Recently, some large corporations have proposed that developers, or a group of developers, set aside wetlands areas and establish new areas that would serve as "wetlands banks". These credits for wetlands creation could be drawn upon in instances where wetlands were destroyed as part of their commercial development activities [Tenneco].

An extension of this concept is to allow the developer to make a money payment to the permitting agency and the agency would then use such money for wetlands replacement. The agency could collect wetlands conversion fees and, when revenues were sufficient, could initiate a wetlands construction/restoration project. Alternatively the agency could construct wetlands and

then collect fees to recover costs. Such a development fee would be set equal to the marginal cost of wetlands replacement. As will be noted below, the fee system may be especially attractive if there exist scale economies in wetlands construction which can only be realized by the management agency. Thus, the second aspect of a wetlands policy reform is to establish a system of fees based upon replacement cost of wetlands services in conjunction with a full mitigation requirement for all wetlands development proposals. As long as the entity causing the damage either provides in kind compensation or pays the fee, the proposed development should proceed. The regulatory problem, once goals are established, becomes one of insuring that the mitigation provisions are appropriate rather than one of trying to assess the benefits and costs of each individual permit prior to making a decision. In this way, the proponents of the development face an opportunity cost based upon a specified and well articulated wetlands policy and they can choose to individually make the adjustments in their development plan which are most appropriate to their own situation.

This policy setting also provides a context for establishing charges for damages to coastal marsh. With the replacement cost policy in place and estimates of replacement cost fees set, there is a basis for charges to be assigned for assessing damages to coastal marsh. To the extent that it can be demonstrated that only a partial loss of wetlands services has been caused (e.g. short term chemical contamination), then the replacement cost fee would be an upper limit on the damage assessment. In addition the practical advantages to the replacement cost approach to damage assessment is that it provides a readily understandable basis for ascribing damages to an environmental alteration. The importance of having generally understandable measurement approaches is argued persuasively by Kimm *et. al.* (1981) in a discussion of benefit-cost analysis in the EPA regulatory process.

> (I)t is predictable that both (or all) sides will be able to then bring forward analyses that support their points of view. Each will then bring forward analysts, of equivalent credentials, who will then argue among themselves about the superiority of their particular analyses in a language all their own. Incomprehensible analysis may then drive out understandable argument [Kimm, *et. al.* 1981, p. 242].

An illustration of the measurement of replacement costs for the Louisiana coast follows in the last section of this paper.

There are three general techniques for the manmade creation of wetlands in Louisiana. These are: controlled diversion, uncontrolled diversion, and the controlled placement of dredge materials. All attempt to duplicate, in some sense, the natural processes that have continued for eons--with sediment laden rivers providing the material for wetlands creation. It is only recently that the construction of dikes, dams, and navigation channels have either trapped sediment before it reaches the mouth of the river or which funnelled the sediment past the delta area to the deep waters of the Gulf of Mexico.

Controlled diversions involve building a structure into the river levee to provide for the controlled release of water through the structure into an area deemed suitable for wetlands. The controlled diversion could be by gravity flow structure, siphons, or pumping stations [Watson, 1984]. The diversion of the flow of the river to restore and to create wetlands has been considered and/or accomplished by the Corps of Engineers and the State of Louisiana [Gagliano and van Beek, 1976; Watson, 1984]. For example, the Violet Siphon Structure was completed in December 1979 in Louisiana's St. Bernard Parish. It consists of two 50 inch diameter pipes which divert a maximum of 250 cfs of Mississippi flow into wetlands lying behind the Mississippi River and Lake Borge [Coastal Environments, 1980]. This flow diversion carries sediment which nourishes existing wetlands and builds new wetlands.

Uncontrolled diversions are similar except that the breach in the levee does not include a control structure. Rather, an artificial crevasse simply diverts some of the river. Since uncontrolled diversion can mean major amounts of river flow through the crevasse during flooding, this technique is usually used well into the delta area, below any population centers.

The third technique used for wetlands construction is the placement of dredged material. Material dredged from navigation channels or sediment rich areas can be pumped into shallow water to create wetlands. Use of dredge material also assists in what otherwise would be a disposal problem, since over 60 million cubic yards of sediment are excavated annually in the Army Corps of Engineers New Orleans maintenance dredging program [Watson, 1984]. Furthermore, dredge

disposal allows for increased flexibility in placement of the wetlands. Whereas, diversions limit wetlands creation to near the river channel, dredge material can be barged or siphoned to alternative areas. Landin estimated that it takes approximately 1613 cy/ft/ac of unconsolidated dredge material to build an acre of wetlands; in Louisiana, the material is placed so that, when it consolidates, the wetlands will be at the intertidal level [Landin, 1986]. Since 1970, over 15,000 acres of wetlands have been created from dredged material in Southern Louisiana. Most of this has been at Southwest Pass and the delta of the Mississippi. Over 4000 acres has also been built using dredged material in the Atchafalaya Basin and other parts fo the Louisiana Coast [Landin, 1986].

The placement of dredge material and the diversion of sediment bearing waters to build wetlands can be considered as a replacement for wetlands destroyed by economic development activities. Therefore, the costs of constructing new wetlands can be used to develop a wetlands management program within an eco-region such as the Louisiana Gulf Coast.

However, computation of the cost of a man-made wetlands area (i.e. a natural wetlands substitute) requires more than simply reporting the engineering cost estimates. This is the case because constructed wetlands may require the passage of time before they can provide the same service level as the natural wetlands area. Landin [1986], for example, estimates that it may require 3 to 5 years for a site in south Louisiana to appear as a natural wetlands; soil profiles and root biomass do not equal that of a natural wetlands until closer to 10 years. However, Landin [1986] further notes that once a wetlands has stabilized--anywhere from 3 to 10 years--there appears to be no functional difference between a manmade and a natural wetlands. Also, engineering costs may include a high initial capital cost component followed by low annual costs over time. Thus, the computation of costs require appropriate adjustments to account for time dimensions of replacement activities. Therefore replacement cost analysis requires the following information for valuing natural wetlands areas:

- What are the recurring annual costs (C) for the wetlands area substitute?
- What is the time period before the substitute can provide a service flow similar to the natural area?

Because the time paths of costs and wetlands service replacement may differ, the analytical problem is to determine the costs replacement. The determination of R, annualized replacement cost, by estimating the value of R which just equals engineering costs incurred for the years during

which replacement actually is accomplished but which includes the time value of money. R can be solved for in the following expression. (Note that R is expressed in annual equivalent terms.)

$$0 = \left[\sum_{t=n}^{T} \frac{R_t}{(1+i)^t} \right] - \left[\sum_{t=1}^{n} K_t(1+i)^t + \sum_{t=1}^{T} \frac{C_t}{(1+i)^t} \right] \qquad (E1)$$

where

T = time period of analysis

t = year

n = year in which substitute provides wetlands service

R = equivalent annual replacement cost of service flow from created wetlands area

i = discount rate

K = costs incurred before substitute begins providing wetlands service (primarily capital costs)

C = recurring annual costs to keep wetlands substitute operating and maintained

In this equation, the present value of the annualized replacement cost of wetland services is shown as the expression in the first set of brackets. The second set of brackets is the actual expenditures for the wetlands substitute. Solving the equation for R yields the equivalent annual replacement cost of the wetlands area.

The general formula expressed above assumes that there are no transitional phases of wetlands development that yield valued services, however, the formula can be easily amended to include such transitional periods of construction.

Case Study--Controlled Diversion

Published technical studies of the potential for wetlands creation in Louisiana provided data with which to estimate the annual equivalent replacement cost. Table 1 describes 7 sites on the Mississippi River where controlled diversions have been studied. At each site, it was estimated that 320 acres of wetlands per year for 7 years could be created by controlled diversion of 25,000 cfs of river flow; construction costs for each site are displayed in column 2. Construction is expected to take two years to complete. Operation, maintenance, and replacement costs were estimated to be 4 percent of construction cost per year. The time horizon for the analysis is 50 years although

wetlands creation ceases at the end of the seventh year. When the formula above is modified to account for the transitional growth of wetlands, the annual equivalent replacement wetlands can be determined for the 7 sites. These costs are displayed in columns (3) and (4). The difference in the estimates arise from the choice of discount rate: 10 or 5 percent.

Case Study-Uncontrolled Division

Uncontrolled diversions in Louisiana involve breaching a mainline levee with cuts varying from 50 to 200 feet wide. [Watson, 1984] These cuts imitate the natural process of river crevasse and overflow. Sediments from crevasses are deposited in low lying areas in deltaic splays. As the delta matures the rate of wetlands created decreases over time.

Table 2 displays areas where potential wetlands creation with uncontrolled sediment diversions have bee studied. Over the first 2 years, a total of 420 acres of wetlands would be created at the 5 sites; however, over 50 years only 5200 will be created. The cuts have to be reopened and extended or relocated every 2 to 3 years to maintain the rate of wetlands rate building [Watson, 1984].

Assuming only one cut was made per site and was not reopened or relocated later, that wetlands building was such that, after 50 years, a total of approximately 5200 acres were created, that there were no maintenance costs after the second year, and that dredging costs are $2.13 per cubic yard, then the estimated annual replacement cost, R, varies from $2.33 per acre to $54.21 per acre at a 10 percent discount rate and from to $1.26 per acre to $29.44 per acre at a 5 percent discount rate (see Table 2).

Table 1
Wetlands Creation With Proposed 25,000 CFS
Controlled Sediment Diversions*

Site Location	(1) Wetlands Created acres/year	(2) Total Construction Cost ($000)	(3) Annualized Replacement Cost Per Acre of Wetlands (10% discount rate) ($)	(4) Annualized Replacement Cost Per Acre of Wetlands (5% discount rate) ($)
Site 1 Mississippi River East Bank-Mi. 34.9	320	$14,019	1244.51	724.37
Site 2 Mississippi River West Bank-Mi. 31.3	320	$27,446	2436.46	1418.14
Site 3 Mississippi River East Bank-Mi. 20.0	320	$16,307	1447.62	842.59
Site 4 Mississippi River West Bank-Mi. 16.4	320	$30,781	2732.52	1590.46
Site 5 Mississippi River East Bank-Mi.	320	$17,914	1590.28	925.62
Site 6 Mississippi River West Bank-Mi. 9.5	320	$19,275	1711.10	995.94
Site 7 Mississippi River West Bank-Mi. 6.0	320	$20,176	1791.08	1042.50
Average Sites 1-7	320	$20,845	1850.51	1077.09

* Modified from Watson, [1984]

Table 2
Wetlands Creations With
Uncontrolled Sediment Diversions*

Site Location	(1) Wetlands Created acres/year	(2) Total Construction Cost	(3) Annualized Replacement Cost Per Acre of Wetlands (10% discount rate) ($)	(4) Annualized Replacement Cost Per Acre of Wetlands (5% discount rate) ($)
Site 1 Octave Pass, South Bank	3.7	115,020	54.20	29.44
Site 2 Raphael Pass, North Bank	3.7	57,510	29.10	14.72
Site 3 Pass a Loutre, North Bank	25	119,300	8.43	4.58
Site 4 Pass a Loutre, South Bank	50	60,000	2.33	1.26
Site 5 South Pass, West Bank	22.5	142,740	11.21	6.09
Average Sites 1-7	21	75,933	21.05	11.22

* Modified from Watson, [1984]

Case Study - Controlled Placement of Dredge Material

The use of dredge material for wetland creation is a favored technique in Louisiana because it is lower cost than other techniques and because it provides an opportunity to use dredge material in a beneficial way. Since the erosion and subsidence of the Louisiana coast is so evident, there is little public opposition to dredge material placement [Landin, 1986].

Landin [1986] estimates that the average amount of dredge material per acre in Louisiana was 4800 cy. In South Louisiana dredging plus dredge material placement averages $1.23/cy. Clearly these costs will be highly sensitive to transport costs. Assuming it takes 3 years to create 2240 acres, that these acres begin to function in the third year, and that there is no maintenance costs after the third year, the value of R annualized replacement cost of services would be $658.89 per acre at a 10 percent discount rate or $345.03 per acre at a 5 percent discount rate.

Summary

During the past decade federal and state government regulation has sought to reduce the rate of coastal wetlands development. However, it is uncertain whether these programs can secure mitigation of damages caused by future development. Revision of coastal wetlands management programs is needed and should involve several actions. First, for regions of concern, wetlands acreage targets would be set; no such targets now guide wetlands regulation programs. As a part of this effort certain areas or types of wetlands might be protected from development. Then, in areas not protected, development would be permitted if the developer paid a fixed wetlands development fee which was set in relation to cost of wetlands replacement. The realized development tax revenues would be used to finance a region-wide investment program which can realize scale economies in replacing most wetlands lost to development. This approach to wetlands management offers more assurance that coastal wetlands damage will be compensated, provides for a more certain regulatory environment for coastal development planning, and provides a policy relevant context for coastal wetlands damage assessment. The case illustration provided for the Louisiana area indicates that replacement costs can be readily computed for various wetlands creation techniques.

The estimated annualized replacement costs was least expensive for uncontrolled sediment diversions (assuming no maintenance costs after 2 years and a gradual building of 5200 acres after 50 years as a result of five different levee "cuts"). The average annual replacement cost per acre when valued at a 5 percent discount rate was $11.22. However, the rate of wetlands created by uncontrolled diversion is slow. After an initial creation of 420 acres over 2 years from the 5 sites, further wetlands growth averages only 5 percent per year.

Dredging material placement, in contrast, would result in annualized replacement costs (R) of $345 per acre, (at a 5 percent discount rate). This R value is, however, very sensitive to transport costs. Nevertheless, the rate and amount of wetlands created are controllable by the rate and amount of dredge material placement.

Controlled diversions were comparably very expensive. Annualized replacement costs averaged $1077 per acre (5 percent discount rate).

These annualized replacement costs can be thought of as the amount of annual fee that would be collected from a permitee if the fee were based on wetlands replacement cost. While the higher annualized replacement costs from controlled diversions, may seem high, consider that, in the first use tax case in Louisiana, state officials were collecting $264 million a year as "fairly related" compensation to wetlands damaged. Use of the expensive controlled diversions to replace the 10,000 acres of damaged wetlands would result in average annual payments from oil and gas producers of only $10.7 million--not overly large considering the returns to be made from oil and gas field developments and much less than $264 million. Use of uncontrolled sediment diversion for wetlands replacement, in contrast would imply average annual payments of only $112,200.

REFERENCES

[Will be provided with later draft]

MEASURING THE ECONOMIC DAMAGES ASSOCIATED WITH TERRESTRIAL POLLUTION OF MARINE ECOSYSTEMS

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PRELIMINARY DRAFT

INTRODUCTION

Marine resources are very often perceived as independent of events that occur on land, but in many instances, particularly for estuarine ecosystems this is not the case. Terrestrial activities, both natural and anthropogenic, have critical implications for marine organisms. For example, the amount of rainfall far to the inland will partially determine the salinity levels of an embayment which will have a direct impact on the size of oyster populations. Of course, anthropogenic activity is going to be more important from a policy perspective. Examples of these activities include the conversion of wetlands, which serve as important nutrient sources for marine organisms and quite often provide nursery and spawning habitat, the terrestrial application of agricultural chemicals, which may reduce aquatic vegetation which will lead to reductions in important fish populations, and diversion of river water for industrial, agricultural or municipal purposes, which may have effects on salinity and nutrient levels of the estuary.

As the above examples suggest, many of the externalities associated with terrestrial activities do not have a direct impact on marine resources or marine environmental quality parameters which are a direct source of utility for man. For example, over a very large range of salinity levels, changes in the salinity of an estuary are not even perceptable to the typical person, even those swimming in the estuary. Rather, changes in social welfare will arise as the effects of the environmental change work their way through the food web or affect natural processes (such as nutrient cycles), having both direct and indirect effects on organisms or environmental quality parameters from which man derives value. In short, the terrestrial activitites often affect environmental goods which are inputs in an ecological production process for what might be regarded as environmental outputs. These environmental outputs are those environmental resources which are utilized by man to produce utility, either by directly consuming the resource (consumption need not be depleting) or by using the environmental good as an input in producing an economic output. This is admittedly an anthropocentric approach, but one which is consistent with the way in which allocation decisions are made

concerning private resources.

For example, assume that policy makers are concerned about the soil erosion from agriculture and its subsequent effects on the population of a marine species that is either commercially or recreationally harvested. Further assume that the fish spawns in an estuarine environment and spends portions of its adult life in both the estuary and the high seas. The estuarine turbidity caused by the inland soil erosion might impact the fish species in many different ways. For example, the suspended sediments might settle out on top of fish eggs, which will deprive them of oxygen. In addition, the turbidity may interfere with the photosynthesis of many different plant species, which will have repercussions throughout the food web, but will ultimately deplete the availability of food for the fish species. Additionally, it could adversely affect the reproductive ability of organisms further down the food chain, or affect their ability to respire, which would ultimately affect the abundance of the fish which is commercially or recreationally exploited.

After even a cursory examination, it becomes apparent that in order to calculate the welfare losses associated with a terrestrial pollutant such as sediment from increased soil erosion, much information must be gathered. First, one must be able to specify the fashion in which inland soil erosion affects the turbidity of the estuary. One then needs to illustrate how the change in turbidity level affects the fish population through all the direct and indirect avenues. In general, this will require a complicated model of the ecosystem which would be commensurate with large scale macroeconomic or regional economic models in terms of their data requirements. Often these data are not available and alternative procedures must be sought. In addition, an economic model of the commercial or recreational exploitation of the fish species and the interaction of exploitation with natural ecological processes must be formulated.

This study suggests methodologies for valuing the damages which terrestrial activitites generate in aquatic ecosystems. While the discussion of these methodologies and their potential application in the valuation process is the primary

focus of this paper, the paper has an additional goal. This goal is to expose natural scientists to the types of models and the kinds of analysis which economists can contribute to the resource managemnt process. This is very important, because, as the paper will illustrate, economists can not operate in this area without input from natural scientists. The nature of the problem implies that at a minimum, economists will need to rely on the counterparts in the sciences to provide critical data on resource populations and other environmental variables. More ideally, economists would work jointly with natural scientists, integrating detailed ecological models with economic models. In the former case, economists must make their counterparts in the natural sciences aware of the type and structure of the data they need, because what is appropriate for ecologic analysis may not be appropriate for economic analysis. In particular, natural scientists who can test their hypothesis by replication across many different field sites may not be collecting data in a sufficient time series to conduct economic analysis. Since the different field sites are likely to span only one market, the economist also will not be able to perform cross-sectional analysis. The data needs will be discussed more fully as the paper proceeds.

The concepts developed in the paper are illustrated by examining the impact of certain terrestrial activities on the productivity of the Appalachicola Estuary in Northwest Florida. Of particular interest is the ability to capture some of these ecological interactions in the absence of a detailed ecological model. The empirical section of the paper will implement some of the methodologies which are suggested in the conceptual part of the paper. The problem which is used to illustrate the methodology is that of measuring the estuarine damages associated with increased upstream water uptake by the growing city of Atlanta, as the estuarine oyster populations are sensitive to the freshwater input from the Appalachicola River.

WELFARE LOSSES ASSOCIATED WITH POPULATION CHANGES

It is clear that in order to be able to estimate the change in welfare associated with a change in the level of a particular terrestrial pollutant, one must be able to specify how the pollutant affects the levels of marine resources which are inputs

into various types of economic activity. For expositional simplicity, it will be assumed that the marine resource is a fish species which is commercially but not recreationally exploited. The extension to recreationally exploited species and to environmental resources (both animate and inanimate) which are inputs to other types of economic activity is quite straightforward, although data might be even more difficult to obtain than the fishery case.

Figure 1 contains a simplified representation of the relationships which must be known before the welfare losses are determined. For the time being, steps (A) through (F) will be ignored, and the discussion will proceed as if these effects are known. The emphasis of the discussion will be placed on how changes in the fish population generate changes in the net social benefits associated with fishing activity. Obviously, a reduced population of fish will generate an upward shift in the supply or marginal cost function. Is is important, however, to distinguish between changes in population generated by the environmental change, and changes in population due to fishing activity (including the response of fishing activity to the new environmental conditions). The mechanism for doing this is the equilibrium growth function[1], which shows the natural growth (F(X)) to a given stock (X) per unit time. In the absence of exploitation, natural growth will equal the net additions to the stock (dX/dt). Both the net additions (and natural growth) and the stock itself are traditionally measured in biomass units rather than in the terms of numbers of individual organisms. Figure 2 shows the equilibrium growth function which is essentially a measure of the productivity of the stock. Note that there are two natural equilibrium levels of the stock, one at a zero level of population, and the other at K, which is the maximum possible level of the population or the carrying capacity of the environment. The growth function can also be interpreted as an equilibrium catch function since population will remain unchanged if catch is equal to F(X) for a given population level. It is this interpretation as an equilibrium catch equation which allows for the separation of the effects of fishing activity and environmental change on the fish population.

One more step must be taken before this uncoupling of the population effects can

be made, and that is to forge a relationship between the growth function (equilibrium catch equation) and the supply function. This is done in Figure 3. Actually, the supply function should not be viewed as a single function, as there will be a different supply function for each level of the fish population. Since each supply function is defined for a specific population level, and the growth function gives the equilibrium catch level defined for each population level, the point on each supply fucntion that is characterized by biological equilibrium (population remains constant) can be identified. LL represents the locus of these points of biological equilibria, which is discussed in more detail in Kahn and Kemp. The level of catch which is characterized by both biological and economic equilibrium is given by the intersection of the demand function, the supply function and the locus of biological equilibria. Net benefits associated with the activity are represented by the consumers' and producers' surplus of area ABC.

A harmful environmental change will shift the growth function downward and to the left (as in Figure 4) and therefore shift the locus of biological equilibira inward as in Figure 5. The loss in social benefits associated with the harmful environmental change is given by the change in consumers' plus producers' surplus, or area EDBC in Figure 5. Note that in the context of this model, one can distinguish between a change in population attributable to environmental changes (a shift of the locus of biological equilibria) and a change in population generated by changes in the level of fishing activity (a movement along the locus of biological equilibria).

The locus of biological equilibria has an appearance similar to the long-run backward bending supply curves discussed by both Copes and Clark. For the purposes of valuing environmental resources it is preferable not to think of supply as a long run equilibrium locus, since the identification of social benefits such as producers' surplus becomes problematic if the supply curve is not truly a supply curve but an equilibrium locus. Each of the supply curves in Figure 3 is capable of identifying producers surplus for the state of the world (level of fish population) associated with that supply curve. This is precisely what one wants in a supply curve as the producers' surplus will adequately reflect the inputs' opportunity costs in the

current state of the world. This difference in interpretation of the supply function appears at first to be rather trivial, but the distinction is very important, especially if one plans to empirically estimate the model. To differentiate the model proposed in this paper from that of the long-run average cost supply model of Copes, this model will be called the contemporaneous supply model.

With its emphasis on the contemporaneously defined supply curve, rather than the long-run supply curve, this study (along with the previous study by Kahn and Kemp which introduced this type of analysis) represents a significant departure from conventional fishery economics. The traditional based yield-effort types of models are quite convenient for deriving the optimal harvest and stock time paths, as well as the maximized fishery rent. However, they are much less well suited for examining benefits other than fisheries rent, such as consumers' and producers' surplus. Of course, if both the marginal cost curve (holding population constant) and the demand curve both approach the horizontal, the contemporaneous supply model loses some of its advantages.[2]

THE IMPACT OF ENVIRONMENTAL CHANGE ON THE GROWTH FUNCTION

As stated above, a harmful environmental change will shift the growth function downward, but the mechanism by which this downward shift occurs has not been discussed. For simplicity, it is assumed that the growth function can be well approximated by the logistic function suggested by Shaefer. This function is presented as Equation 1, where K is the carrying capacity of the environment, r is the intrinsic growth rate and X is the population of the organism.

$$F(X) = rX\left[1-\frac{X}{K}\right] \qquad (1)$$

A harmful environmental change can impact on the growth function in two ways, either by changing r, the intrinsic growth rate, or K, the carrying capacity. For example, the massive infusion of DDT into the environment in the 1950s and 1960s

probably diminished r for birds of prey such as eagles and ospreys, but not K. The ability of the environment to support eagles and ospreys (K) was probably not significantly diminshed by DDT (although it may have been by other factors, such as loss of habitat). However, the presence of DDT in the raptors' bodies is believed to have interfered with their reproductive ability. This would serve to decrease r since the intrinsic growth rate is the maximum proportionate growth rate or the proportionate growth rate when the population is very small. Even though there may be abundant food and habitat, F(X) will fall. Alternatively, an environmental disturbance such as the destruction of habitat or pollution which reduces fish populations upon which the raptors prey will serve to reduce K which will in turn reduce F(X).

The effects of reducing r and K are not completly symmetric. For an unexploited population, a decrease in r will diminish F(X) but not lower the long-run equilibrium value of X. This can be seen by looking at equation (2) which is the solution to the differential equation (1) (when the population is unexploited F(X) = dX/dt). On the other hand, a decrease in K will lower both F(X) and the maximum population which X can obtain. Of course, in the presence of exploitation, the reduction in r will imply either a reduced population if effort is held constant or a reduced level of exploitation if population is to remain constant. The corresponding solution for the differential equation dX/dt when fishing effort (E) occurs is found in equation (3).[3]

$$X(t) = \frac{K}{1 + \left[\frac{K-X(t_0)}{X(t_0)}\right] e^{-rt}} \qquad (2)$$

$$\lim_{t\to\infty} X(t) = K(1-E/r) \qquad (3)$$

COMPUTATION OF WELFARE LOSS IN THE CONTEXT OF AN ECOSYSTEM MODEL

For the time being, the ability to estimate an ecosystem model will be taken as given. An ecosystem model can be defined as a model which specifies (as a system of differential equations) the flows of mass or energy within the ecosystem, subject to the constraints of the system. Such constraints are both external to the organisms in the ecosystem (salinity levels, the availability of dissolved oxygen, nutrients, *etc.*) and internal (metabolic rates, reproductive potential, *etc.*). Although a discussion of the issues involved in estimation of such a model is beyond the scope of this paper, it is possible to estimate such a model using a combination of field and laboratory data. This process would be analogous to estimating cost functions based on engineering data.

If the parameters of the model can be estimated, the ecosystem model can be solved for given values of the exogenous variables. The solution will consist of equilibrium population levels of the various organisms, which can be used to derive the carrying capacity of the species of interest. If the system is unexploited, the carrying capacity is the equilibrium level. If the system is exploited, (and the level of exploitation known) the carrying capacity can be identified. Hence, it is possible to forge a link between the the exogenous environmental quality variables and the carrying capacity of the species of interest.

This link provides the ability to identify the marine welfare losses stemming from terrrestrial pollution, provided the demand and supply functions (or inverse demand and marginal cost functions)are known for the economic output for which the marine species is an input. Inverse demand and marginal cost functions can be specified as in equations (4) and (5), respectively. Π represents the inverse demand for catch, MC represents the marginal cost of catch, C represents the catch (in biomass terms), ρ_i represents a vector of input prices, ρ_s represents a vector of prices of subsitute commodities, and S represents a vector of socioeconomic variables. Equation (6) contains the natural growth function of equation (1)

interpreted as an equilibrium catch equation.

$$MC = MC(C, \rho_i, X) \qquad (4)$$

$$\Pi = \Pi(C, \rho_s, S) \qquad (5)$$

$$C = rX - (r/K)X^2 \qquad (6)$$

Given these functions, the economic equilibrium condition that MC=Π and the ecosystem model it is possible to estimate the changes in net social benefits associated with an environmental change. The ecosystem model may be solved assuming a baseline environment to yield baseline values of K and r, which can be plugged into the equilibrium catch equation. The right hand sides of equations (4) and (5) are then set equal to each other (fulfilling the equilibrium condition) and the right hand side of equation (6) substituted into the resulting equation. This equation will be a quadratic equation in X, for which the solution is the equilibrium population level (X^*). C^* can be determined by solving equation (6) for the catch level, which is an equilibrium catch in both the biological and economic sense. The net benefits associated with this commercial fishing activity can then be calculated according to equation (7).

$$\text{net benefits} = \int_0^{C^*} [\Pi - MC]dC \qquad (7)$$

The welfare losses associated with the environmental change can then be calculated by solving the ecosystem model with the new levels of the environemental parameters. This will return new values of r and K, which can again be used to solve the system of equations (4)-(6). The new values of C^* and X^* are then used in re-evaluating equation (7). The difference between the pre-change level of net benefits and the post-change level of net benefits represents the welfare change associated with the environmental change. Note that changes in environmental variables will alter net benefits through two avenues. Changes in the environmental variables will change the equilibrium population and thus shift MC, and it will also affect the level

of C^*.

THE MEASUREMENT OF WELFARE CHANGE IN THE ABSENCE OF AN ECOSYSTEM MODEL

The above discussed model is predicated upon the existence of a comprehensive ecosystem model. If this model has in fact been estimated for the ecosystem of interest, the calculation of the welfare loss associated with environmental change is relatively straightfoward. Unfortunately, such an ecosystem model will not be available in most cases. The data requirements for this type of model are such that an extensive and expensive data collection effort must be made over a relatively long period of time. Very often policy decisions will have to be made in a more timely fashion, without the luxury of first estimating an ecosystem model. However, if certain types of data are available, it is still possible to determine the welfare losses associated with an environmental perturbation. This can be done in a straightforward fashion provided that data on the population of the critical species, the catch of the critical species, and appropriate environmental quality variables are available. The methodology is outlined below.

The natural growth function of equation (1) is easily estimated when written in the following form

$$F(X_t) = rX_t - (r/K)X_t^2 + \epsilon_t = \alpha_1 X_t + \alpha_2 X_t^2 + \epsilon_t \qquad (8)$$

where $F(X_t)$ is the recruitment to the stock in the year t and can be measured as $X_t - X_{t-1} + C_{t-1}$, where C_{t-1} is the catch in year t-1.

An examination of the error term of this equation can lead to some important insights into the direction in which the analysis should procede. In a completely constant environment, equation (8) would be essentially an identity, with any residual due to measurement error alone or possible small random fluctuations systematically unrelated to any variable in the system. Of course when equation

eight is estimated using time series data, the error term is much more encompassing than this, as the assumption of a constant environment is not valid and the error term will embody actual flucuations in K (or r) due to an environment which varies over time. Such environmental changes would include naturally generated changes due to factors such as weather, or from man-made causes such as pollution or destruction of habitat. The flucuations in K which are generated by changes in environmental parameters are exactly what must be explained in order to begin determining the benefits or costs associated with the changes in the environmental parameters. If one can obtain field observations from a series (cross-sectional or time) of unexploited subsystems of the ecosystem, then the estimation of K as a function of environmental parameters such as salinity levels, nutrient concentrations, dissolved oxygen levels, *etc.* is quite straight forward. In general this may prove difficult to do, because if the species of interest is really important, few if any unexploited ecosystems are likely to exist.

There is an approach which can work quite well, however, as long as the data on population, catch and the environmental variables are available. This approach would be to estimate equation (3), but rather than include the unobserved K on the right hand side, substitute some function of the environmental variables. If Q_k represents the k^{th} environmental variable, then equation (3) can be rewritten as equation (9).

$$F(X) = rX\left[1 - \frac{X}{\sum_{k=1}^{n} \delta_k Q_k}\right] \qquad (9)$$

where F(X) is approximated as $X_t - X_{t-1} + C_{t-1}$[4]. Note that the denominator of the fraction is presented as a linear function. This is done for analytical simplicity, but a nonlinear function could be easily substituted.

Equation (9) should be easily estimable using nonlinear estimation techniques. The natural growth function specified in equation (9) can then be interpreted as an

equilibrium catch equation and substituted for equation (6) in the system of equations which describes the bioeconomic equilibrium for the economic activity. The inverse demand and marginal cost functions are presented as linear functions in equations (10) and (11). The three equation system ((9)-(11)) can be solved as before, and net benefits calculated in the same fashion,. The only difference is that the intermediate step of computing K has been eliminated. The original values (or naturally occuring values) of the various Q_k can be subsituted into equation (9) and the model solved to yield the net benefits associated with the economic activity. The level of a particular Q_k can then be varied and the model solved again to find the welfare losses associated with a change in Q_k.

$$MC = a_0 + a_1C + \sum_{i=1}^{m} \alpha_i \rho_i + a_3X + \theta \qquad (10)$$

$$\Pi = b_0 + b_1C + \sum_{j=1}^{h} \beta_j S_j + \phi \qquad (11)$$

Very often the ability to estimate such a model will be limited by the number of observations on population. For a variety of reasons, data on the species population is likely to be available for a much shorter period of time than for the economic variables and many of the other environmental variables. In some circumstances the time series may be so short that there are insufficient degrees of freedom to estimate equation (9). However, as long as a reasonable number of observations on population exist, it is possible to construct a fairly reliable proxy for population. This proxy is based on one already suggested in the literature, catch per unit effort. However, rather than use catch per unit effort as a proxy for X directly in the supply function (under these circumstances one would not even attempt to estimate the growth function), catch per unit effort is used to provide an estimate of population for those years for which catch per unit effort data is available, but not population data. This can be done by running a simple regression (equation 12) of population on catch per unit effort. If catch per unit effort is as good a proxy for population as has

been suggested by the frequency of its use in the fisheries literature, then $\hat{X}$ should be a good approximation for X for those periods in which C and E are observed, but not X.

$$X_t = \gamma_0 + \gamma_1 \left[\frac{C_t}{E_t} \right] + \psi_t \qquad (12)$$

In many circumstances, there may no observations on population, or simply too few observations to even attempt estimating the simple regression of equation (12). Under these circumstances, the attempt to estimate the equilibrium catch equation becomes problematic [5] and a modified and less satisfactory methodology must be employed. This methodology would involve collapsing the three equaition system of equations (9)-(11) into the two equation system of equations (13) and (14). Note that the major modification has been to eliminate the unobserved X from the right hand side of the marginal cost function and substitute those environmental variables (Q) which partially determine X. With this system it is still possible to measure the change in net benefits associated with a change in a particular Q_k, however caution must be employed as the results are likely to be a reasonable approximation only for marginal changes in Q_k. The reason for this is that when the biological equilibrium is no longer explicitly considered in the solution of the overall equilibrium, an implicit assumption of biological equilibrium must be made. As changes in the environmental variables (Q) displace the equilibrium further and further from the original equilibrium, the assumption that the population remains in biological equilibrium becomes increasingly strained.

$$MC = a_0 + a_1 C + \sum_{i=1}^{m} \alpha_i \rho_i + \sum_{k=1}^{n} \alpha_k Q_k + \theta \qquad (13)$$

$$\Pi = b_0 + b_1 C + \sum_{j=1}^{h} \beta_j S_j + \phi \qquad (14)$$

Both the three equation model (equations (10)-(12)) and the modifed two equation

model (equations (13) and (14)) model will be estimated for oysters in the Appalachicola Estuary in Northern Florida. The environmental variables which appear to be important in the equilibrium catch function are the flow of the Appalachicola River and the area of oyster reef. Oyster reefs are important because the oyster spat must fall on hard bottomed areas in order to survive. Oyster reefs are naturally occuring phenomena which can be augmented by the planting of discarded shells from oyster shucking operations. The proportion of total bottom area covered by reefs is largely independent of environmental variables which can be influenced by policy, but dependent on truly random events such as hurricanes which may cover existing reefs with sediment, rendering them incapable of supporting oysters. River flow is important because it is linked to the salinity of the estuary, which is hypothesized to have a nonmonotonic (increasing and then decreasing) effect on oyster populations (Meeter, *et al.*, Chanley). As Atlanta (which is located upstream on the Apalachicola River) grows, a greater volume of water is removed to service the population and industry in the region. The three equation model can measure the welfare losses associated with the decline in oyster populations associated with the diversion of river flow, both at the margin and in total.However, the two equation model can only estimate marginal changes in net social benefits.

(*NOTE TO DISCUSSANTS: The two equation model will be estimated in time for the workshop in June. Unfortunately, the data on osyter populations will not be available until late summer or early fall, so I will not be able to present those results, as I had hoped.*)

CONCLUSIONS

It is possible to estimate the welfare losses associated with terrestrial pollution of marine ecosystems if the effect of the environmental change on the parameters of the equilibrium growth function can be specified. The ideal way to do this is by integrating the economic model with a detailed ecosystem model, but these detailed ecosystem models are often unavailable. In the absence of such a mdoel it is possible to specify the welfare losses provided that data on the environmental

variables and the populations of critical species are available. This is done by including these environmental variables as explanatory variables in the equilibrium catch equation. When it is not possible to estimate an equilibrium catch equation (no population data) it is still possible to approximate these welfare losses by including the environmental variables as explanatory variables directly in the marginal cost functions. This model, however, is less satisfactory because it does not explicitly model the biological equilibrium of the species of interest.

An important area of future research is to test these (and competing) methodologies in as many different applications as possible. However, the ability to engage in this type of analysis is constrained by data limitations. In order to develop better policy, it is neccessary to develope more and better data. Of particular importance is time series data on the levels of species' populations and environmental quality variables which are of policy relevance.

Figure 1.

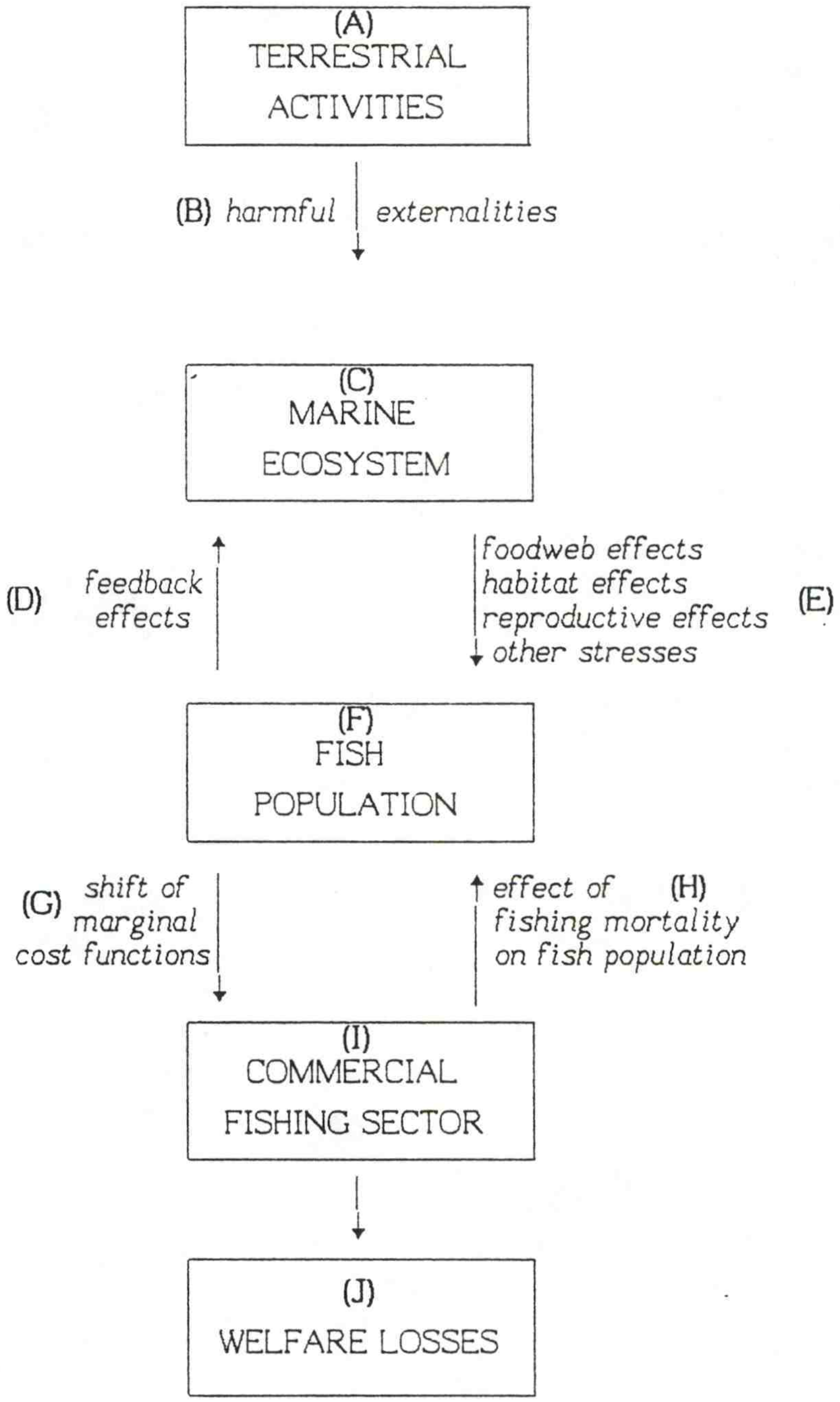

(A)
TERRESTRIAL
ACTIVITIES
(B) harmful externalities
(C)
MARINE
ECOSYSTEM
(D) feedback effects
foodweb effects
habitat effects
reproductive effects
other stresses
(E)
(F)
FISH
POPULATION
(G) shift of marginal cost functions
effect of (H)
fishing mortality
on fish population
(I)
COMMERCIAL
FISHING SECTOR
(J)
WELFARE LOSSES

Fig 2

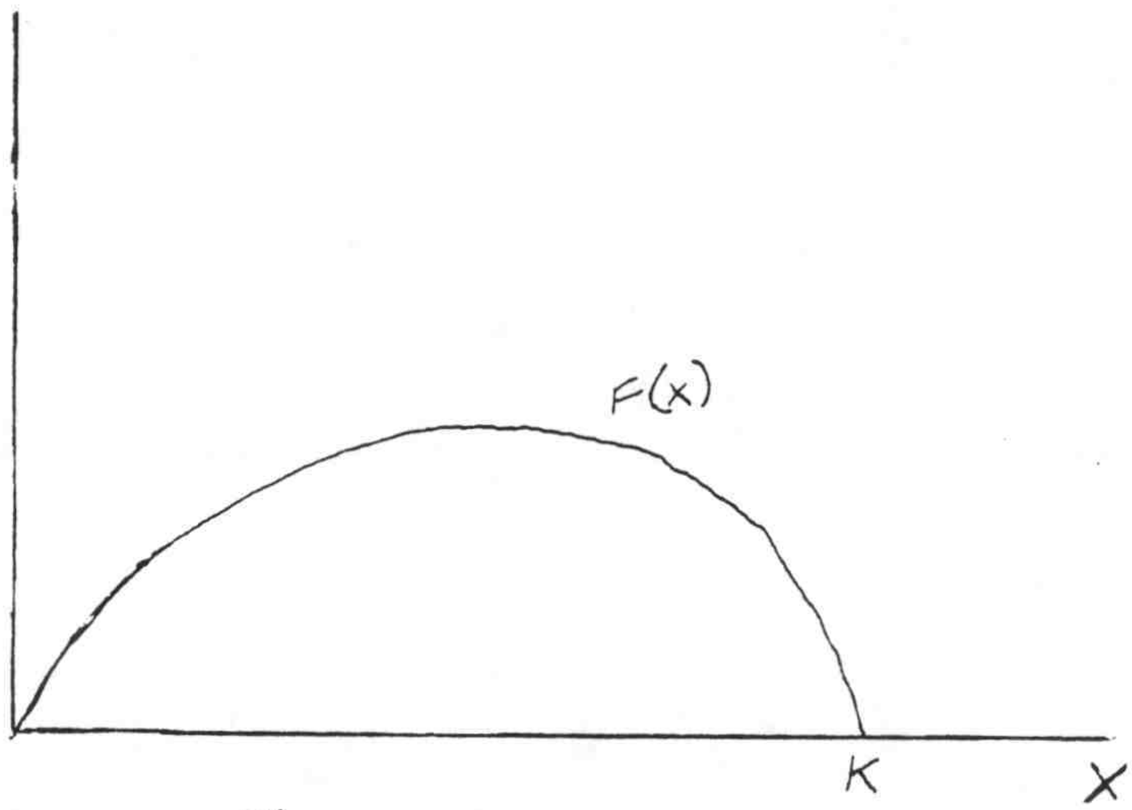


Fig 3

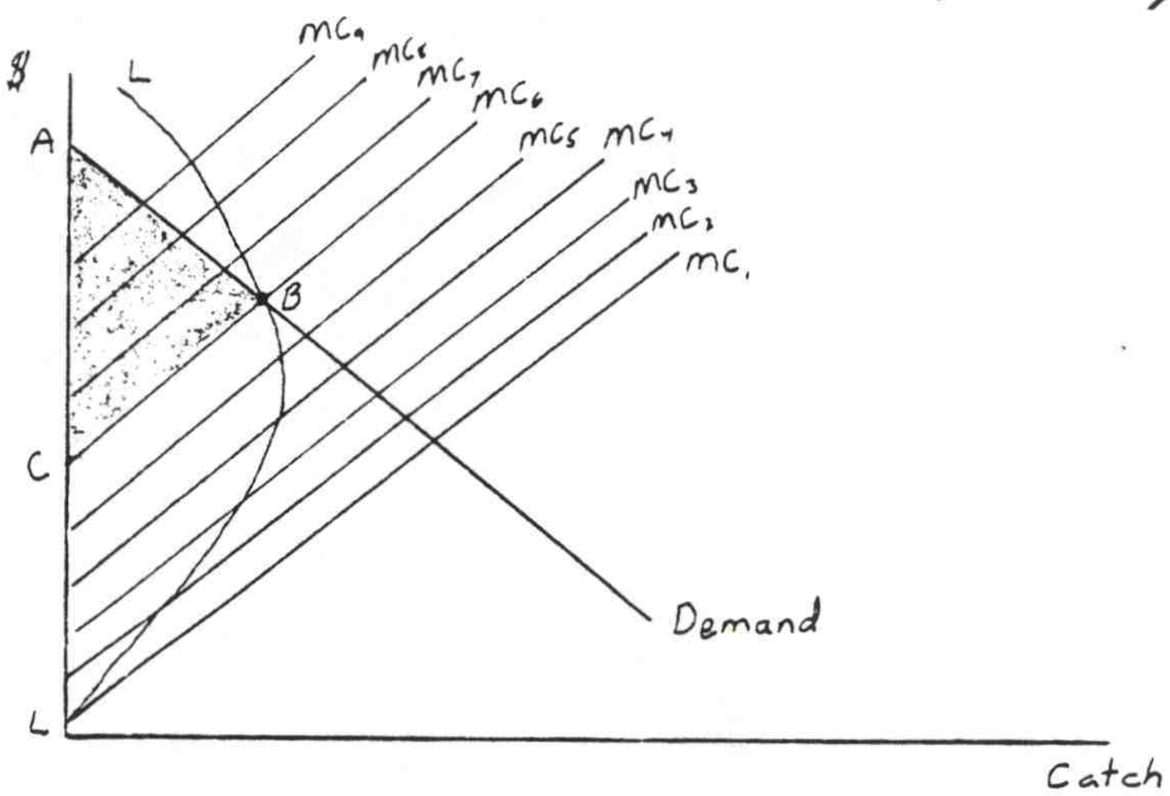


Fig 4

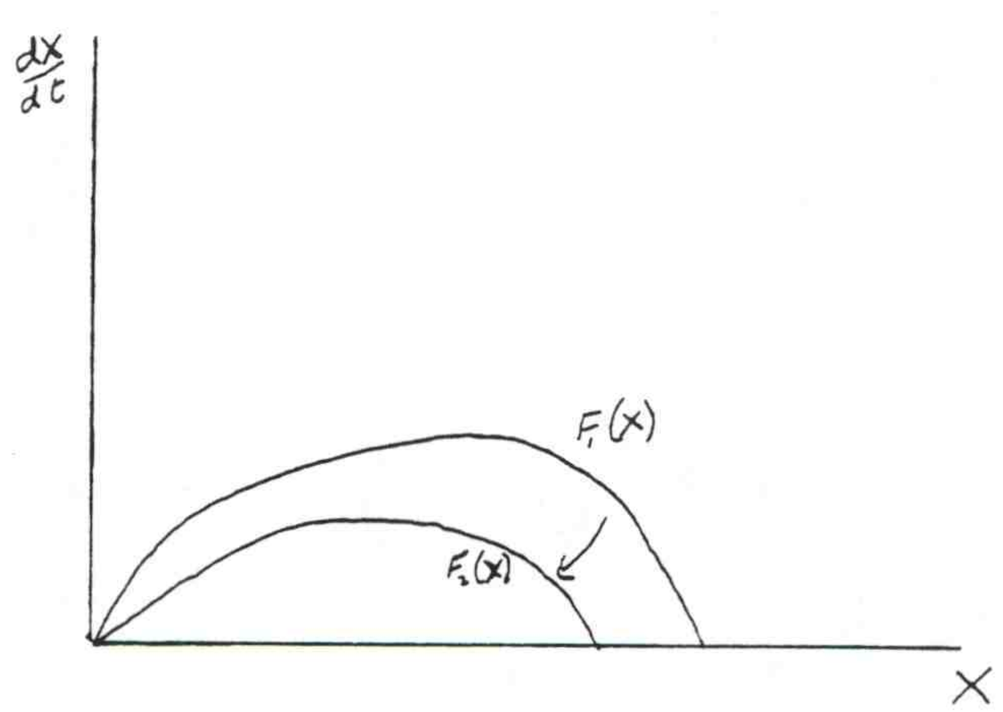
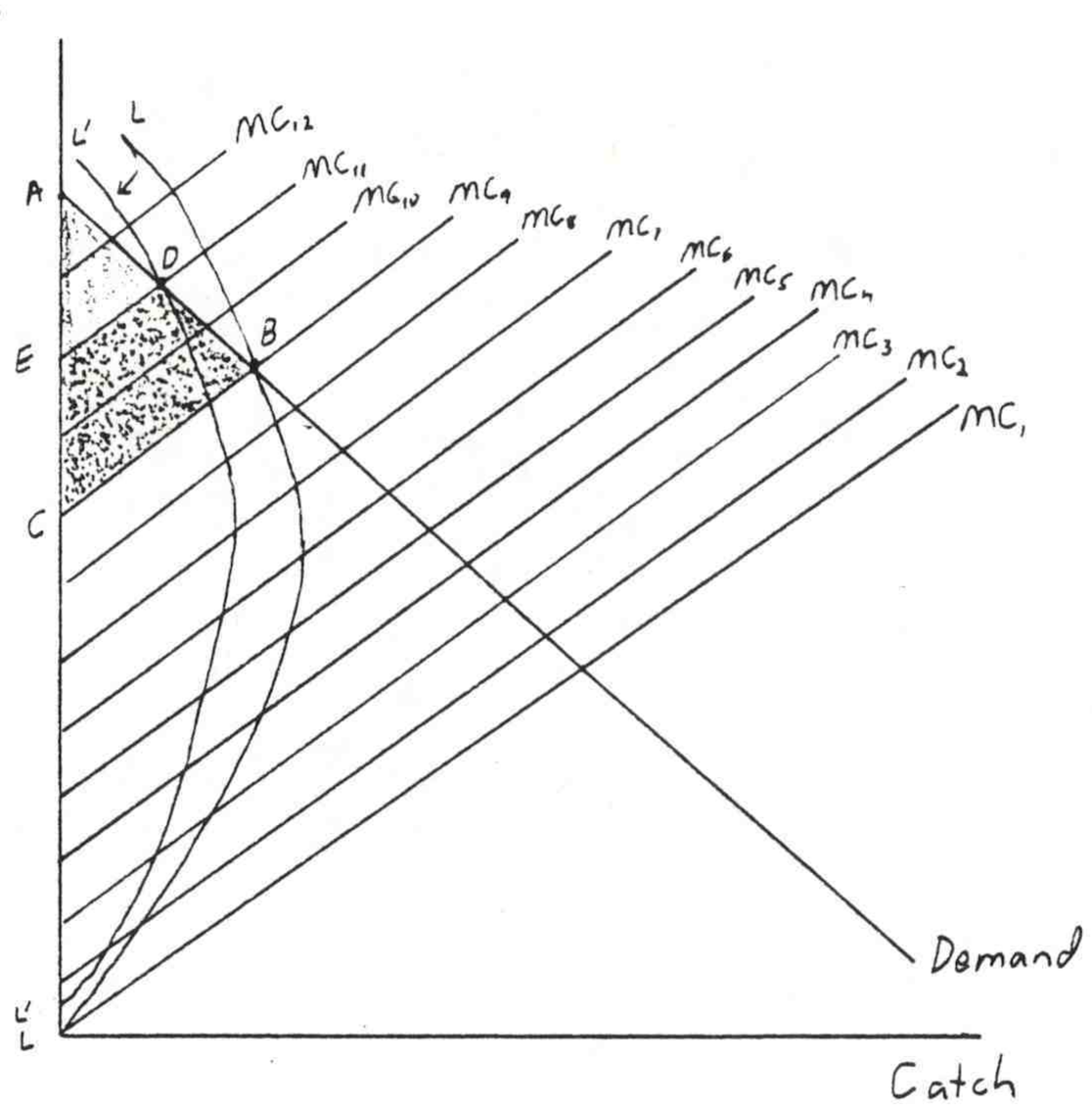


Fig 5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MC_3
MC_2
MC_1
Demand
Catch

FOOTNOTES

1. See Clark for a detailed discussion of the growth function as well as for citations of some of the seminal studies which investigate it.

2. Another important advantage of the contemporaneous supply function model over the yield effort model can be found when the models are empirical estimated. The yield effort model requires the definition of an effort variable, and the definintion of effort can be quite arbitrary. Boats and gear of different types must be somehow expressed in a common metric, and these capital inputs must also be aggregated together with other types of inputs, such as labor and energy. The contemporaneous supply model does not require such aggregation, because rather than using units of the inputs as an explanatory model, it employs input prices. These input prices can be entered separately and the problem of aggregating across input categories is avoided. There remains the issue of aggregating within an input class, for example in nearshore fishing, some boats use regular gasoline and some boats use diesel gasoline, so there is some question of how to construct a price variable for the energy input. Empirically, this is not really an important problem, however, as long as the prices of different types of fuel tend to move in tandem.

3. The derivation of both equation (2) and equation (3) are presented in Clark.

4. For certain species an age-class structure of population may be more appropriate than a total biomass apecification.

5. Chambers and Strand offer a methodology for estimating growth functions when population data is missing. Under certain circumstances, their procedures may be adoptable to the above outlined methodology. However, since their procedures rely heavily on the use of lagged variables and since time series are likely to be short in many of the applications to which the present paper is addressed, there may be insufficeint degrees of freedom to employ their procedure.

SESSION III. Conceptual Needs for Marine Pollution Policy

ECONOMIC VALUATION OF MARINE WILDLIFE: DOES EXISTENCE VALUE EXIST?

by

Ronda K. Hageman

Abstract: Yes.

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Crucial to the implementation of the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA, 1980) is development of viable and appropriate methods for assessing natural resource damage in polluting incidents. Regarding marine ecosystems, damage assessment methodology must address the increased potential for the occurrence of damaging incidents affecting marine resources as coastal development proceeds. Only some marine resources are transacted in commercial markets so that only a portion of societal values lost are revealed through market values. Though it is generally recognized that additional societal losses associated with damage to non-marketed biological systems should be evaluated, ongoing discussions under the CERCLA have yet to determine how such values should be quantified.

As evidenced by strong public support for the Marine Mammals Protection Act (1972) and various incidents such as Alaskan residents' protest of the live capture of non-endangered killer whales in 1984, society places some value on preservation of marine wildlife. Clearly, optimal policy choices are hindered by the uncertainty associated with the likelihood of damaging incidents, the extent of their effect on marine wildlife, and society's value associated with such effects. The following discussion will address this last issue in a presentation of results of a contingent valuation study conducted to determine Californians' economic values for protected marine mammals. Analysis of results will incorporate a discussion of non-market valuation issues which must be addressed if wildlife valuation is to be meaningful in a policy context.

Previous authors have argued convincingly for consideration of total economic value of natural resources, of which market value is often just a subset (see, e.g., Randall and Stoll, 1983, Boyle and Bishop, 1985). In this regard, two methodologies -- the travel cost method and the contingent valuation method (CVM)-- have been used in recent years in a large number of environmental quality and recreational settings. In the context of public policy, benefits estimation of public natural resource programs has taken on greater importance in the current era of fiscal responsiblitly; thus, use of both methods has received greater attention and acceptability. For example, explicit measurement of recreational/aesthetic benefits is required by the Water Resources Council (unit day values), the U.S. Forest Service's Resource Planning Act (values for hunting and fishing), and the Bureau of Land Management's Rangeland Investment Policy Act. And, under the Marine Mammals Protection Act (Section 2(6)), marine mammals are "resources of great international significance, aesthetic and recreational as well as economic, and it is the sense of Congress that they should be encouraged to develop to the greatest extent feasible commensurate with sound policies of resource management" (emphasis added).

However, very little research has been conducted to quantify the non-market benefits associated with wildlife. Of the few studies which exist, almost all have been directed at valuing consumptive uses; i.e., the recreational values for hunting and fishing. The travel cost method, especially, has been refined considerably through a large number of applications (see, e.g., McConnell and Strand (1981), Miller and Hay (1984), and Huppert and Thomson (1984)). Though these studies have demonstrated the feasibility of deriving consumer surplus values associated with hunting and fishing, these measures do not include non-consumptive values which may

exist. This is probably a non-issue for many managed fish stocks and animal populations, but it may be a problem for estimating valuations for marine wildlife where hunting and fishing are either disallowed or not desired, and even sitings are not common.

The contingent valuation method has been applied in a very small number of studies in order to elicit estimates of individuals' values for animal species. The CVM study cited most frequently is perhaps that of Bishop and Heberlein (1979) in which the consumer surplus value for goose hunting in Wisconsin was investigated. Another CVM study conducted by mail was reported by Brookshire, Eubanks, and Randall (1983). This study asked hunters to reveal willingness to pay for future hunting permits for grizzly bear and bighorn sheep. Hunters were also asked for non-use values, i.e. willingness to pay to observe the animals, but Stoll and Johnson (1984) conducted the first willingess-to-pay survey of non-hunters for a protected wildlife population. This study elicited values for whooping cranes which can be observed at the Aransas Refuge in Texas. In two recent studies, Boyle and Bishop (1985) and Hageman (1985) investigated non-market valuations for wildlife populations which are neither hunted nor are they always observable in the wild. What follows is a discussion of results from the latter study.

Before proceeding, it is useful to outline the types of benefits for which marine mammals are valued. Mendelsohn (1984) has compiled a comprehensive list of relevant benefits which is presented here without elaboration. These are: consumptive and non-consumptive recreation, indirect recreation by way of media exposure (films, books), bequest value, "chemical mining," research on chemicals and genetics, experimental value, pest control, enhancement of other desired species (i.e., importance in the food chain), option and quasi-option values, and existence value. After a generally persuasive discussion of each, Mendelsohn argues that use values only are relevant for measurement of the benefits of preserving endangered species. The utilitarian argument is that all other non-use values, such as option value or existence value, are in fact use values which are being double-counted in benefits elsewhere. For example, the argument is posited that existence value does <u>not</u> exist, and that if people were allowed <u>no</u> information on the animal stock (precluding not only visits, but also media information), then willingness-to-pay for blind faith in the animals' continued existence would be zero.

We take issue with the strictly utilitarian approach to value measurement for marine mammals and other animal populations. It <u>may</u> be true that existence value is zero when it is narrowly defined to preclude all direct and indirect exposure to the animals or information about the animals, but this is strictly conjecture. Even if we accept the conjecture, the question arises as to how, then, are total use values to be measured? Mendelsohn argues that these values are capture in payments for movies, television documentaries, live zoo and aquarium exhibits, books, and artwork. However, an effort to enumerate the large number of multi-media exposures for any particular animal and to thus estimate the total willingness-to-pay for that species would generally be such an enormous task as to render it an impossible endeavor. Furthermore, conversations with individuals frequently reveal that at least some people adamantly claim their values are <u>not</u> tied to utilitarian concerns. Even if we concede that such individuals actually value animal species because of what might be defined broadly as a

utilitarian concern for ecological integrity which is necessary for the human species' long-term survival, we would argue that the issue is one of semantics.

For the purposes of the study described in the next sections, existence value is defined as the maximum willingness-to-pay for those benefits which are not tied to direct use (neither consumptive nor non-consumptive). By direct non-consumptive use, we are referring to current or future on-site observation of animal populations. The distinction is important because it allows individuals to indicate their values even though current or future uses are not intended. In this way, damage assessment for detrimental effects of marine pollution on wildlife need not be tied necessarily to losses in observation/recreation opportunities. In the section which follows, the CVM survey structure is described and results are provided which speak to potential biases discussed in the CVM literature. Also, valuation estimates of California households are reported for four marine mammal populations. Differences between values are discussed, and several issues relevant for the appropriate application of such value estimates are addressed. For example, do households have a specific value for each animal population versus marine mammals (or perhaps ecosystems) in general? Can households provide information on the value they attach to <u>incremental</u> changes in wildlife populations? If sampled households can provide valuations for losses of marine wildlife, are the responses representative of the affected population in the event of marine pollution, and what is the appropriate population over which to aggregate? For marine wildlife, does existence value exist?

APPLICATION OF THE CVM IN MARINE MAMMALS VALUATION

Analysis of the usefulness of the CVM for marine mammals valuation is based upon the results of a survey mailed from San Diego State University in 1984. The sample population was 1,000 California residents. Names and addresses were randomly chosen from telephone books according to the population distribution of the state (based upon the U.S. Census of Population 1980) - 21.9% were were sent to San Francisco/Oakland/San Jose, 48.6% to Los Angeles/Long Beach/Anaheim, 7.4% to San Diego County, 3.6% to Sacramento, 9.7% to other urbanized areas, and 7.6% to rural areas (places with less than 2,500 residents.)

<u>Survey Description</u>

Following Dillman (1978), the survey procedure consisted of three mailings. The first included an introductory letter which outlined the purpose of the survey and assured confidentiality, a brief description/ directions sheet, two descriptive sheets on the mammal groups to be evaluated, a questionnaire, two yellow answer sheets, and a self-addressed stamped return envelope. The second mailing was a reminder postcard sent to those households from which responses had not been received. The third mailing was another letter accompanied by a second copy of the survey materials.

The four species of mammals which were described to the respondents are representative of marine mammals in California. All surveys requested

responses for bottlenose dolphins, California sea otters, and northern elephant seals. However, half of the surveys also asked respondents to provide responses for gray whales, whereas half were asked to answer for blue whales. The first whale population is quite abundant and can be viewed easily on whale watching tours or from the coast, whereas the second is quite rare (some researchers believe the blue whale population to be beyond recovery) and virtually impossible for anyone but researchers to observe. These four particular species were chosen in order to represent a spectrum of attributes - appealing versus unattractive (sea otters versus elephant seals), visible versus inaccessible, large versus small, familiar versus unfamiliar, endangered versus non-endangered.

In the descriptions of the four species, the following information was provided:

(1) A typical picture of one or more animals in the wild. Appealing pictures or textbook drawings were avoided in order to approximate a typical viewing experience.

(2) A small map indicating the range of each population.

(3) A scale of population levels and dates at which they have occurred in the past. For all mammals, the following scale applied:

A. a best estimate of the undisturbed population, before human activity. This scenario was dated to show when excessive hunting of the animal began off the California coast.

B. an incremental increase in the population above current level C, but below the historical maximum A.

C. the 1984 population level which exists under protective legislation.

D. a population level which reflects a best estimate of the historical low number of animals when hunting was allowed. This "no protection" case was dated to provided information on what happened to the populations when hunting was unregulated. (This was not relevant for dolphins since the California population has not been hunted.)

(4) <u>History</u>: A brief discussion of the animals and information on on whether or not they have been considered to be in danger because of human and/or natural causes.

(5) <u>Current population</u>: Estimates of the number of animals off the California coast and how these populations are changing.

(6) <u>Worldwide</u>: How the California population compares to the worldwide numbers of these animals. For example, are there many other animals of this same type found around the world? Or are most or all of this type of animal found along California's coast?

(7) Seeing the animal: How accessible the animals are for viewing and and photographing in the wild. The respondent was referred to the map which illustrates range. Also, some rough figures were provided on the average number of animals per square mile of ocean near the shoreline within the range for each of the situations A-D. The respondent was advised to use this information to get some idea of his/her chances of seeing the animals.

The questionnaire itself was divided into three parts--travel cost information, the CVM study, and socio-economic questions. In Part I, titled "Seeing the Animal," a brief orientation was provided in the introductory questions which ask respondents to report exposure to a species through communications media, captive display, or actual observation in the wild. (According to Dillman, as a means of encouraging respondents to continue, the first questions should be ones which require little effort and which will have "yes" answers for most people.) The remainder of this part attempted to identify travel behavior for respondents who reported recent observation in the wild; those who did not report recent observation were referred to Part II.

Part II, titled "Importance of the Animal," used the CVM to elicit valuation responses from both users (i.e., observers) and non-users. The responses to Part I and Part II were recorded on the first page of the yellow answer sheets. The respondents reread the questions in Parts I and II four times, answering all questions for whales first, then for bottlenose dolphins, California sea otters, and northern elephant seals. The answer sheet was divided into five columns. The first column gave brief instructions for each question, and each of the other columns provided answer spaces for the same questions asked for each of the four species. At the bottom of the answer sheet, a payment " bid card" (as suggested by Carson and Mitchell (1984.) to allow respondents to focus on their bid without creating starting point bias) was provided for use in the WTP questions. Payment choices ranged from $0 to $200, with low values incremented by small amounts. Values from $20 to $100 were incremented by $5, and over $100 by larger amounts.

The second and final page of the answer sheets was entitled "About You." On this sheet, individuals were asked to provide confidential information on socio-economic variables: number of residents in the household, age and sex of respondent, employment and annual income, years of education, and whether or not hunting/fishing or membership in an environmental group applied to adult members of the household. Also, an "Avidity Scale" was described, on which respondents were asked to indicate their avidity on a 0-10 scale for each of the following: swimming, sailing, surfing, sunning at the beach; ocean activities which require a motorized boat; fishing for sport (shellfish and billfish) in the ocean; protection of ocean animal populations; protection of any animal population if endangered; and preservation of "wilderness" types of areas where no human development or machinery are allowed.

In the past, many valuation studies have utilized face-to-face interviews to collect data. However, budget constraints have led

PART II. IMPORTANCE OF THE ANIMAL

Please answer the following questions whether or not you have seen this animal in the wild or elsewhere. Some people believe that hunting (if allowed), pollution and fishing nets in the ocean could destroy many marine mammals. Some people even believe that without protection these animals might not survive in the ocean off the California coast. This animal is protected by government programs which, of course, have costs. The following questions are designed to find out how much your household values protection of this animal.

9. Please look at the chart shown on the left-hand side of the Description Sheet for this animal. The level marked C shows the current population size. Assume for a moment that this animal is no longer protected from hunting or other types of damage. Assume also that without protection, the population would fall to Situation D. This would, of course, decrease your chances of seeing the animal and could also endanger the population. Please look over the descriptions about the animal as you think about moving from C to D. Suppose that the only way of avoiding Situation D is if households were willing to contribute to a fund specifically used for this purpose. Suppose also that each household in the nation were required to pay the average amount of all households' answers to the following question, rather than the actual amount of your response. What would be the maximum amount (in dollars) your household would be willing to pay per year into the fund to protect this animal and prevent Situation D? PLEASE CHOOSE YOUR ANSWER FROM THE PAYMENT CHOICES SHOWN AT THE BOTTOM OF THE YELLOW ANSWER SHEET.

10. Suppose a survey such as this was conducted, but the average responses to Question 9 did not provide enough funds to prevent Situation D. Please look at the payment choices at the bottom of the yellow answer sheet and indicate any additional amount over and above your response to Question 9 which your household would be willing to pay at most per year into the fund to prevent Situation D.

11. Your maximum yearly payment is found by adding together the numbers you gave in Question 9 plus Question 10. Please write this total next to #11 on your answer sheet.

The questionnaire was structured to avoid several potential problems. The discussion below addresses several areas about which criticisms of the CVM have been raised due the potential for bias.

Strategic Behavior

To reduce the likelihood of strategic behavior in the reporting of valuations, an incentive-free payment mechanism was introduced. Respondents are asked to report annual WTP for their household, given that the actual payment would be an average of all respondents valuations. Furthermore, to encourage true revelation of preference and avoid free riding behavior, the stipulation was made that all individuals would be required to contribute this amount in the hypothetical situation.

Even so, some strategic behavior could still exist on the part of individuals who strongly favor or disfavor the public good. Therefore, one reason for collecting socio-economic data later on in the survey is to enable us to identify outliers when the results are analyzed. This allows for some control on the few respondents who may attempt to behave strategically. Rational individuals who are not behaving strategically may be expected to report a maximum WTP which reflects perceived benefits at the margin. Perceived benefits, or utility can be expected to be a

researchers to turn to mail and telephone survey techniques instead. Dillman (1978) describes a number of tested techniques which not only help to insure that interview bias does not result, but also enhance response rates. This survey incorporated many of these techniques; for example, each introductory letter was personally addressed and hand signed. Also, though budgetary limitations prohibited offering respondents any financial incentives for filling out their questionnaires, some incentive was provided by promising to send respondents a copy of the study results. However, Dillman also suggests the use of a booklet form for the questionnaire, ideally with an appealing cover illustration. Unfortunately, printing costs made this impractical for this CVM study. Because several species were of interest, a booklet would have been quite large if questions were repeated several times so that answers could be made on the booklet. Thus, it seemed more reasonable to use one questionnaire for all species, and employ one answer sheet which also made clear the idea that four different species were being valued but the approach was the same in all cases.

Methodological/Theoretical Issues

The initial contingent valuation questions are shown below. The individual is referred to the description sheets for the animal population and asked to state a willingness-to-pay amount to avoid moving from the current Situation C to Situation D. The payment vehicle of an earmarked fund was chosen to avoid the negative connotations which generally attend tax payments. Since we are referring to free roaming animal species, a user fee did not seem to be an appropriate payment vehicle, especially since it was not expected *a priori* that all households would have observed or plan to observe all of the species in the wild.

function of income and perhaps other variables. For example:

> U(income, location of residence, family size, age, occupation, education, previous exposure to marine mammals or other wildlife, avidity for marine recreation and/or wildlife conservation, etc.)

If econometric analysis indicates that an individual's WTP deviates significantly from the reported WTP of individuals with similar socio-economic characteristics, this may indicate strategic behavior and this data point can be removed (see Section V for a detailed discussion of this procedure).

However, as noted previously, strategic behavior is rarely identified in CVM studies. (For example, in a CVM study by Brookshire in which campers were asked to state their WTP to preserve a recreational site, the only case of strategic behavior appeared to be an economist who happened to be vacationing at the site with his family.) Generally, we would expect people to have little incentive to report biased WTP due to the hypothetical nature of the questions.

Hypothetical Bias

The willingness-to-pay questions were structured to provide as much consequence realism as possible. Population characteristics (threatened/endangered status accessibility for viewers, uniqueness, range) were described in order to determine if respondents take such information into account when stating valuations for different species of mammals.

The first question, willingness-to-pay to avoid deterioration from the current Situation C to Situation D was based upon historical evidence of drastic reductions in the respective marine mammal populations when protection was neither funded nor enforced. To further encourage participants to provide considered WTP responses, individuals were asked to consider all monthly expenses (utilities and home expenses, entertainment, food and clothing, education, or charity) when making a final valuation estimate. This was included to counter a criticism of the CVM that individual's values are estimated in partial equilibrium.

The willingness-to-pay response to prevent deterioration from C to D can be depicted graphically using indifference curve analysis, as shown in Figure 2. If environmental degradation (reduction in marine mammal populations) is depicted along the horizontal axis, and (B_1, G_1) represents an individual's current position on indifference curve I_0, then the maximum WTP to prevent deterioration to X_1 is Y_1-Y_0, an equivalent variation measure (EV_D). In this case, the property rights do not rest with the respondent. However, the respondent may also be asked to state a willingness-to-pay to obtain an improvement from X_0 to X_1. The compensating variation measure (CV_I) will be equal to EV_D for the same change in environmental quality. Thus, it is valid to state the willingness-to-pay question in terms of either the deterioration or improvement situation since the magnitude of the stated value will be the same in either case.

Payment Vehicle Bias

In order to avoid protests due to use of increased tax payments as a payment vehicle, individuals were asked to state WTP into a preservation fund to be used to protect marine mammals.

PART II. IMPORTANCE OF THE ANIMAL

Please answer the following questions whether or not you have seen this animal in the wild or elsewhere. Some people believe that hunting (if allowed), pollution and fishing nets in the ocean could destroy many marine mammals. Some people even believe that without protection these animals might not survive in the ocean off the California coast. This animal is protected by government programs which, of course, have costs. The following questions are designed to find out how much your household values protection of this animal.

9. Please look at the chart shown on the left-hand side of the Description Sheet for this animal. The level marked C shows the current population size. Assume for a moment that this animal is no longer protected from hunting or other types of damage. Assume also that without protection, the population would fall to Situation D. This would, of course, decrease your chances of seeing the animal and could also endanger the population. Please look over the descriptions about the animal as you think about moving from C to D. Suppose that the only way of avoiding Situation D is if households were willing to contribute to a fund specifically used for this purpose. Suppose also that each household in the nation were required to pay the average amount of all households' answers to the following question, rather than the actual amount of your response. What would be the maximum amount (in dollars) your household would be willing to pay per year into the fund to protect this animal and prevent Situation D? PLEASE CHOOSE YOUR ANSWER FROM THE PAYMENT CHOICES SHOWN AT THE BOTTOM OF THE YELLOW ANSWER SHEET.

Information Bias

Given our desire to encourage a higher response rate by limiting the survey's length, we provided as much information on each species as one-half page would allow. This included the historical setting, population status and location, potential for siting, and a picture of the animal. Every effort was made to avoid making sympathetic statements about endangered species or to show aesthetically appealing pictures/scenes which would not be viewed in actual sitings in the wild.

Protests

Aside from individuals who wrote to say they could not or would not respond, we also used the following question to identify protest bids:

12. On the first yellow answer sheet, please circle the answer from the choices below which best describe your reason for responding to Questions 9 and 10 as you did.

A. CHOSE BEST ESTIMATES OF WHAT SHOULD BE PAID TO PREVENT SITUATION D.
B. DO NOT FEEL WE SHOULD PAY, BUT THE GOVERNMENT SHOULD.
C. THE FUND DESCRIBED IS AN INAPPROPRIATE WAY TO PROTECT THIS ANIMAL.
D. COULD NOT AFFORD ANY MORE.
E. UNWILLING TO ESTIMATE DOLLAR AMOUNTS EVEN THOUGH HOUSEHOLD VALUES THIS ANIMAL.

Those respondents who stated a zero WTP, and also answered "B", "C", or "E" were identified as protestors and were removed from the sample.

ANALYSIS OF RESULTS

In response to the first mailing and follow-up reminder cards, 121 questionnaires were completed and returned. After the final mailing in which a second copy of the questionnaires was enclosed for individuals who had not yet responded, a total response rate of 21% was achieved. Of this total, eleven were identified as protests either by the response to the control question or by the respondent's written explanation. (This included some, but not all, of the zero WTP responses received). Fourteen individuals either misunderstood the questionnaire or filled out only portions of it.

There remain 180 usable responses for whales (93 gray and 87 blue), 175 for dolphins, and 174 for sea otters and elephant seals (i.e, some individuals filled out their questionnaire for only some species, leaving the others blank).

Contingent Valuation Responses

Missing values for all variables except the WTP data were replaced by estimates derived by using the modified first order regression method for estimating missing observations. Madalla (1977) reports this is the preferable method when the correlation between variables is less than 0.5. A preliminary inspection of the correlation matrix without missing observations indicated low correlation values. The value of the maximum WTP per year per household (question #11) adjusted by reported values for #14a (adjustment to WTP after discussing income constraints) is used as the final WTP estimate. Means and standard deviations for WTP were calculated for each of the four species, by survey group (blue whales or gray whales in the first column). The eight groups were viewed separately at first to determine whether the effect of having blue whales versus gray whales resulted in statistically different responses for each of the four species. Mean values with standard deviations are reported in Table 1.

Using Student's t-tables, equality of means by species was tested with the following results (t-statistics, degrees of freedom are shown in parentheses below):

H_o: Gray Whales = Blue Whales (-0.237,178)

H_o: $Dolphins_G = Dolphins_B$ (-0.167,173)

H_o: $Sea\ Otters_G = Sea\ otters_B$ (-0.162,172)

H_o: $Elephant\ Seals_G = Elephant\ Seals_B$ (-0.21,172)

In all cases, the null hypotheses are accepted at a confidence level greater than 99%. Data were then pooled into four groups, by species.

Since some previous analyses of the results of mail surveys have provided evidence that responses from follow-up mailings do not affect the results derived from responses to initial mailings (see Goudy 1978 and Wellman et al. 1980), we compared the WTP responses and the answers to socio-economic questions from the first 121 respondents (Data Set I) to the responses received after the final mailing (Data Set II). Because there is no reason to believe a priori that the two data sets are not independent samples, a t-test was performed on the difference in means for each species in the early data set and late data set. The results shown in Table 2 indicate that the responses received after the final mailing were not statistically different (confidence interval exceeding 99%) than those received earlier. Furthermore, responses to socio-economic characteristics in Data Set I were compared to those in Data Set II. For every variable, a t-test on the difference in means indicated that the socio-economic characteristics of early respondents were not statistically different than the characteristics of later respondents. However, in order to decrease the chances of making a Type I or II error, the entire sample was used for analysis in the discussion which follows.

Mean responses to the initial willingness-to-pay question (#9) are shown in Table 3 for the four species. These values reflect the initially stated annual WTP per household. In order to investigate the bidding behavior of respondents, t-statistics were estimated, by species, to determine if the mean response to question #10 is significantly different from zero. This question asked the respondent to state any amount he/she would be willing to pay over and above the WTP stated initially in order to assure the present situation (marine mammal protection) as opposed to the no protection scenario. The t-statistics are reported in Table 4. Furthermore, a similar test was performed on responses to #14a, the bid adjustment after the respondent's income constraint is discussed as follows:

14. Consider for a moment your household's budget. Some of the expenditures which you are currently making would have to be reduced if you made your payment in Question 11 to prevent Situation D. With this in mind, would you like to revise your payment into the fund?

If NO, skip to Question 15.
If YES:

14a. Looking at the payment choices on the bottom, how much would you like to revise your payment? (For example, +50¢/year or -$1.00/year.) Please continue to Question 15.

15. If your household is still choosing to pay some amount into the fund for this animal, which of the following budget categories would you reduce in order to pay into the fund?

A. Utilities and home expenses
B. Entertainment
C. Food and clothing
D. Educational expenses
E. Charity contributions to other causes

Inspection of Table 4 reveals that the original willingness-to-pay is, in fact, much smaller than maximum willingness-to-pay (#9 and #10) (elicited by the question: "Suppose...the average responses...did not provide enough funds to prevent Situation D...Please indicate any additional amount over and above your [initial] response...") Previous studies (Schulze, Brookshire et al. (1983), Desvousges et al. (1983) and Burness et al. (1983), for example) have observed the same result, so that we conclude that the "bidding process" is important if the CVM is to provide evidence on maximum WTP. Furthermore, though the introduction of the household budget (Question 14) did result in some apparent decrease in WTP for each species, except sea otters, it turns out that these adjustments were not statistically different from zero. This result has been observed in previous studies (Burness et al. 1983 and Schulze, Brookshire et al. 1983.) It can be viewed as some evidence that individuals provide considered information on their valuations to the preceding maximum WTP question under contingent conditions. In Cummings et al. (1984), it is suggested that finding this result is evidence that the application of CVM indicates that the WTP response is a "preference-researched bid" rather than a random number, and that income/commodity trade-offs were considered by respondents when they offered responses. Although this is not a complete counter to the issue of hypothetical bias, it does provide some evidence of introspective reporting of individuals' WTP values for marine mammal protection.

In order to include the preferences of those respondents who made adjustments to the WTP estimates, the adjusted value (Q11 $\pm$ Q14a) is retained for the remainder of this analysis. The correlation matrices for WTP, by species, and the socio-economic variables are shown in Tables 5, 6, 7 and 8. The following variables are defined as:

EXP = 1 + 2 + 3 + 4

AVM1 = (AV26 + AV27 + AV28)/3

AVM2 = (AV29 + AV30 + AV31)/3

These are indices, where EXP represents exposure to the mammal through the media, captive display, and on-site observation. This is the sum of the respondent's answers to Questions 1, 2, 3, and 4, where yes = 1 and no - 0. The two average measures of avidity, AVM1 and AVM2, are indices of enthusiasm for marine recreation and wildlife/nature conservation, respectively. AVM1 is the average of responses on the 0-10 scale in Questions 26-28; AVM2 is the average of responses on the 0 - 10 scale in Questions 20 - 31. Zero represents no avidity and 10 represents extreme avidity.

Inspection of Table 5 provides some information about the impact of the socio-economic characteristics on maximum WTP. Exposure to the mammals, avidity for marine recreation, and membership in environmental groups have very low, positive correlations with the willingness-to-pay responses. Mileage to the coast (the horizontal distance from city center of residence to the coastline) has an extremely low, negative correlation with WTP. Avidity for wildlife/nature conservation is somewhat more correlated with WTP, but the value though positive, is still quite low. Even income and education have fairly low correlations with WTP (though presence of outliers could be affecting this relationship.) Age is negatively correlated with WTP possibly due to the impact on WTP of responses by retired persons on fixed incomes.

Before proceeding with a discussion of the maximum WTP estimates, we must address the common criticism that some individuals who strongly favor or disfavor the public good being valued may have attempted to bias the results when reporting their WTP values. Even if we argue that the hypothetical nature of the study reduces the incentive for such intentional behavior, it is this hypothetical nature which could instead cause individuals to mistakenly misstate their true willingness-to-pay. A way to reduce these possibilities is to identify probable outliers in the data set and remove those responses. One method might be to simply eliminate observations which lie some X (say 10) standard deviations from the mean. However, this adjustment to the sample seems rather arbitrary and does not allow for any consideration of the respondent's characteristics (which could affect her/ his stated WTP) relative to other respondents in the sample.

In this study, identification of likely outliers is accomplished by using a diagnostic technique suggested by Belsley, Kuh and Welsch (1980). Use of the technique requires first, regressing explanatory socio-economic variables on the WTP estimates for all observations. In CVM studies, application of ordinary least squares (OLS) regression analysis to estimate WTP has generally yielded R^2 values of 0.3 or less using cross-sectional data. The reason these regressions have little explanatory power is that utility functions which determine values for public goods tend to be highly individualized. In our example, we might hypothesize that residents nearer to the coastline would value marine mammals more highly than inland residents. However, there may be many inland residents who value the mountains and undeveloped nature, so that they too report a high value for species in general, including marine mammals. Thus, we would expect a variable like miles from the coast to have a negative coefficient when regressed on WTP, but it may not have a strong or even statistically significant effect.

$$WTP = f(EXP, MC, FSZ, AGE, Y, AV2)$$

where:
- WTP = Q11 ± Q14a
- EXP = exposure to the animals through the news media, captive display, or on-site observation
- MC = mileage of town of residence to the California coast
- FSZ = family size; number of residents in the household
- AGE = age of respondent
- Y = annual household income
- AVM2= avidity index for species preservation/conservation

The other avidity type of variable, such as membership in environmental organizations and AVM1 were not included because they did not appear to affect WTP. Furthermore, they may be measuring the same effect as AVM2. The education variable was not included because, on theoretical grounds, it is too closely associated with income and could cause multicollinearity.

All data except miles to coast were taken directly from the returned answer sheets. Miles to coast were determined to be the horizontal map distance from the respondent's town of residence to the California coastline. A priori, we would expect exposure, income, and avidity to have a positive effect on WTP, and inspection of Table 6 reveals this to be the case. Family size decreases WTP, perhaps because it lowers per person income. (Inclusion of Y/FSZ rather than Y reduced the explanatory power of the equations.) As age of the respondent increased in this data set, WTP was reduced. This could be the result of having several retired individuals in the data set on fixed incomes. The coefficient on mileage to the coast has a negative sign, but it is never a significant variable based upon the t-statistics shown.

As shown in Table 6, removal of likely outliers resulted in an improved R^2 and also a statistically significant coefficient on income, as we would expect intuitively. Although the high standard error which results from the use of a cross sectional data set such as this does not allow use of these regression results to predict bids, the procedure does allow the removal of likely outliers, which enhances the reliability of the WTP estimates derived from the remaining data. In Table 7, a profile of the likely outliers is shown. For purposes of comparison, mean values for the remaining data set are provided in Table 8. Below, an index is provided for the questions for which mean response values are shown in Tables 7 and 8.

However, if an OLS regression has some theoretical justification (e.g., income has been shown to affect WTP in many previous CVM studies), it can be used to identify outlying observations. Belsley, Kuh and Welsch have developed a statistic which essentially re-estimates the coefficients in the WTP equation sequentially without each observation. If an observation significantly changes the coefficient, that response is identified as a likely outlier. This technique has been applied in two previous CVM analyses, (Desvousges et al. 1982 and Brookshire et al. 1984). It seems an essential step since the possibilities of strategic behavior or hypothetical bias could result in incorrect valuations reported by some respondents.

After performing an OLS regression on the maximum WTP responses for each species, we calculated (nxm) B-K-W statistics, one for each variable and each respondent on a particular species. Because economic theory supports the notion that WTP should be determined, to some extent, by income we used the B-K-W statistics on income as our gauge to identify outlying observation. Following Desvousges et al. (1982) and Brookshire et al. (1984), the B-K-W statistic for a particular observation divided by the regression coefficient on income exceeded 0.3, the observation is labeled a likely outlier. The interpretation is that the B-K-W statistic indicates that this observation alone caused a change in the coefficient on income in excess of 30%. The 30% gauge was a natural cut-off point in this study, since almost all coefficients were affected by less than 20% for all variables.

For all species, the same two individuals' responses were identified as outliers. In addition, a third respondent was identified as an outlier in the dolphin and sea otter data sets. In other studies utilizing this technique, some outliers were identified which had WTP values very near the mean (i.e., if a respondent's stated WTP was extremely unusual given his/her socio-economic make-up relative to similar types of respondents); however, in this study the outliers identified were, in fact, only the very high bids received. (For example, one respondent who bid $400/year wrote to say that the individual was strongly in favor of wildlife conservation, but expressed concern that similar households would respond by over-estimating true WTP.)

In Table 6, results of the OLS linear regression procedures are shown for the independent explanatory variables before identification of outliers and then after outliers have been removed from the data set. Based upon a preliminary inspection of the correlation matrices, the following variables were included:

The socio-economic profile of survey respondents is similar to that of average Californians. Based upon 1980 census figures, average household size is 2.68; for our respondents, the average is 2.67. Average age of the respondents is about 42 years; this compares to an average of 43.5 for the adult (over 19) population of California, as reported in the California Almanac (Fay et al, 1984). Average income per family for 1984 in California is $32,602/year (again, inflating 1980 Census figures to 1984 dollars.) Therefore, the average income of the survey respondents, approximately $35,000 per year, is near that of the general population of the state. Average education of respondents is 15.3 years (± 2.9), compared to a statewide average of 12.4 years in 1980. Again, the survey respondents exhibit a similarity to the general population of California.

The information gathered on avidity shows that the mean response to the questions about enthusiasm for marine recreation activities (AV26 - 28) is at or below the mid-point of 5. Mean avidity responses for wildlife/ wilderness preservation (AV29-31) were above the midpoint but below the maximum. In order to make comparisons between respondents' and average Californians' avidity for marine recreation/resources, a telephone survey was conducted. An independent sample of 425 California households was chosen, distributed over all areas of the state population of California. Respondents to the telephone survey answered questions only related to avidity, as shown below:

Hello,

My name is_____. I'm a student at San Diego State University, and I'm working on a project to find out how Californians feel about ocean resources and recreation in our state. If you don't mind, I'd like to ask your opinion on six questions. It will only take two or three minutes of your time.

Picture a scale from 0 to 10 on which you can rank your avidity (desire or enthusiasm) for the things I'll describe. 0 means no avidity. 10 means extreme avidity. 5, or course, is something in between the two extremes.

On this scale, please give me the number from 0 to 10 you'd choose to represent your avidity for:

Q-1 Swimming, sailing, surfing, and sunning at the beach, 0 to 10?
Q-2 Ocean activities which require a motorized boat, 0 to 10?
Q-3 Fishing for sport in the ocean (for example, shellfish and billfish), 0 to 10?
Q-4 Protection of ocean animal populations, 0 to 10?
Q-5 Protection of any animal population if it is endangered, 0 to 10?
Q-6 Preservation of wilderness types of areas, 0 to 10?

That's the last question. Thank you very much for your time.

No mention of marine mammals was made so that responses would reflect general avidity for the activities/issues discussed. However, the six questions asked were identical to the last six questions (26-31) on the mail survey. Eighty-three percent of the households called were at home; of these, 71% answered all six questions. The following averages from 250 Californians' responses were obtained.

	Telephone Survey		Mail Survey	
	Mean	Standard Deviation	Mean	Standard Deviation
AV26	7.1	2.9	5.5	3.2
AV27	4.9	3.4	2.6	3.1
AV28	5.3	3.6	2.0	2.7
AV29	8.8	1.9	6.9	2.7
AV30	9.2	1.7	7.3	2.6
AV31	9.1	1.5	7.8	2.6
AVM1 (Average of 26, 27, 28)	5.8 1	2.6 1	3.4 1	3.0 1
AVM2 (Average of 29, 30, 31)	9.1	1.5	7.3	2.6

Hypothesis tests for equality of means between the two surveys indicate that the mean avidity values reported in the mail survey are not statistically greater than the mean values for avidity stated in the telephone survey (greater than 99% confidence for all questions). This result provides further evidence that the respondents who mailed their valuations for marine mammals are no more avid about marine resources or environmental protection than the typical California household.

Data on the miles to coast variable were not provided by the respondents. These were calculated by estimating the horizontal distance from city centers of residents in the sample to the California coastline. At the outset of the study, every effort was made to draw survey names from cities and towns around the state based upon the total population distribution. For the surveys mailed, the miles to coast from city centers averaged 21.8 miles, largely due to the fact that 78% of the population lives in the San Francisco, Los Angeles, or San Diego Areas.

Although information on the average distance of residence from the coast for all California residents is not readily available, we were able to tabulate what proportion of the population lives within 100 horizontal miles from the coast. It turns out that 95.7% of Californians reside in cities whose centers are 100 horizontal miles from the coast, whereas 99.1% reside within 130 miles from the coast. This is relevant for our study because over 12% of the respondents who returned completed questionnaires lived in excess of 70 miles from the coast, and maximum mileage was 130 miles (3 respondents.)

In Table 9, the means and standard deviations of the maximum WTP responses are shown, stated by respondents as the amount per household per year. These are the values <u>after</u> likely outliers have been identified and removed using the B-K-W procedure discussed above. For purposes of comparison, the values for the entire data set are shown in parentheses. When outliers are removed from the data set, mean WTP estimates decline somewhat with a rather dramatic decrease in their standard deviations. This reduces the coefficient of variation (standard deviation divided by expected value) by 22% for whales, about 33% for dolphins and elephant

seals, and 50% for sea otters.

The overall mean WTP across all species is $20.21. Since most respondents, 171 out of 178, provided WTP estimates for all four species we can pair those responses by species and calculate t values to determine if the differences in responses from one species to another were significantly different from zero. The results are shown in Table 10. (Since the same respondents answered for all four species, these four mean values were not drawn from independent samples, and so a t-test on the difference of means would not be valid.)

These results indicate that respondents appear to have made some distinction in reporting WTP for different species. The difference in mean values reported for whales and sea otters is not significantly different from zero, but the difference between whales and dolphins or elephant seals is significantly greater than zero. Likewise, the difference between mean WTP for sea otters and elephant seals is significantly greater than zero, and approximately so for the difference between the means of WTP for sea otters and dolphins also. However, the difference between mean WTP for dolphins and elephant seals is not statistically greater than zero.

There might be several reasons for the evidence of some statistically significant differences in WTP between species shown in Table 10. It might be argued that gray whales and sea otters are easier to observe in California and therefore may have more non-consumptive use value. However, blue whales were generally not seen by respondents, yet mean values for them were statistically the same as for gray whales. Also, public exposure to bottlenose dolphins is probably as great, since the wild population lives within a few hundred yards of southern California beaches and the popular dolphin shows at oceanaria also use bottlenose dolphins. Thus, if WTP were attributable largely to " cuteness" and "intelligence", it would seem that bottlenose dolphins would rank at least as highly as sea otters.

One difference which may explain the relatively higher mean valuations for whales and sea otters could be current population status, as described in the species information sheets. Sea otters are a threatened species and are found in California in very small numbers; the same is true for blue whales. This is not true for gray whales, though heightened public awareness about the past endangered condition of the species off California's coast may affect public values regarding this species.

CONCLUSIONS AND CAVEATS: SOME THOUGHTS ON APPLICATION OF CVM RESULTS

Program Benefits: Aggregating over Affected Households

In Table 9, evidence is provided from the CVM study that the average willingness-to-pay per household in the sample is $23.95, $17.73, $20.75, and $18.29 for protection of the current populations of gray and blue whales, bottlenose dolphins, California sea otters, and northern elephant seals, respectively. Although the payments for whales and sea otters are statistically greater than payments for dolphins and elephant seals, indicating that respondents value these species differently, it might not be true that this implies a total average WTP of $80.72 for all four mammal populations (the sum of the individual averages). Kahneman (1984) has suggested that because of their inexperience in making direct payments for environmental goods, individuals may be drawing upon an "environmental account" in the case of each stated WTP.

Along these lines, we might reason that since the initial instructions to the individual explained that the purpose of the survey was to elicit public valuations for marine mammal protection programs, then some basic amount is budgeted to the "marine mammal protection account" (say, for example, an average amount of $10), and additional amounts represent the respondent's willingness-to-pay for protection of the specific species of mammal discussed. The sum of these marginal benefits from protection of each species would then be the maximum willingness-to-pay for all marine mammals protected.

However, even if this were the case, we do not have information on the proportion of the WTP estimates reflecting the general "marine mammal account." Thus, to avoid over-estimating societal benefits attributable to marine mammal protection programs, we will use only one WTP estimate in aggregating over all California households. Assuming the respondents' average WTP of $23.95/year per household for whales is representative of average Californian households, and including only households in cities within 100 horizontal miles from the coast (95.14% of the total), we arrive at the following measure of program benefits to Californians:

$$\text{Annual Aggregate Benefits} = \$23.95 \times (23{,}667{,}902/2.68).9514$$
$$= \$201.23 \text{ Million (1984 dollars).}$$

This estimate of annual program benefits, slightly greater than 200 million dollars is an aggregate for California households where the 1980 Census of Population for California is divided by 2.68 persons per household. This measure of benefits is for Californians only. It may be that residents of other states also benefit from marine mammal protection, but only a national CVM study would determine the average value of national WTP.

The reliability of this estimate of aggregate benefits in California depends upon first, the existence of bias in the WTP estimates, and second the extent to which average respondents' values represent average Californians'. With respect to bias, every attempt was made to encourage

informed responses by furnishing information about the marine mammals being valued. Furthermore, accepted techniques were utilized for discouraging strategic behavior. To further reduce the effect of strategic and/or hypothetical bias, likely outliers were identified and removed. Evidence is presented that individuals had considered income/commodity trade-offs when stating their maximum WTP values because they did not significantly adjust their bids when given the opportunity to re-evaluate within their income constraint. Also, individuals' behavior was in accord with theoretical precepts; i.e., the WTP values shown in Table 11 for increments to the marine mammal populations are diminishing, as we would expect for situations of decreasing scarcity.

A problem with mail surveys of this type is the low response rate. While surveys mailed to special interest groups generally attain response rates of 75% or greater, this type of survey must, of necessity, be directed at a random sample of the population. As a result, CVM researchers mailing questionnaires to a random population rarely attain response rates in excess of 35%. This may lead to questions about the representativeness of the responses relative to the average individuals; for example, if only overly concerned individuals returned their questionnaires, then the WTP averages may be upwardly biased. In this study, inspection of the socio-economic characteristics in Table 8 leads us to posit that respondents are in fact, representative of average Californians. This position is further supported by the fact that respondents' avidity for marine recreation and environmental issues was no stronger than avidity rankings provided by Californians in an independent telephone survey.

Appropriate appplication of CVM results must address the issue of the relevant population over which aggregation is performed. In this estimate, we included most of the population of California since our survey responses were returned from a large spectrum of areas, urban and rural, north and south, and beach and non-beach communities. Most importantly, there was almost no correlation of the stated willingness to pay with the respondents' distance of residence from the coastline. Thus it seems reasonable to aggregate over 8.4 million of California's 8.8 million households. Furthermore, it may be that residents of other states also benefit from marine mammal protection, but a national CVM survey was not conducted. However, in their whooping crane valuation study, Stoll and Johnson (1984) found that option price/existence value reported by out-of-state non-users was 75% - 100% of the option price/existence value reported by in-state residents.

Valuing Individual Species of Marine Wildlife

In some instances, damages to an individual species may require valuation of one animal population. For example, the California sea otter is listed as a threatened species due to its susceptibility to potential oil spills in the marine environment. Again, based upon Kahneman's (1984) environmental account framework, we might posit that respondents in a CVM

The Existence of Existence Value

The results in Table 12 have important policy implications. Even when there has been no "use," individuals still explicitly stated that option price/existence value associated with marine mammals is 15.2 times as great as use value for northern elephant seals, 9.3 times as great for blue and gray whales, and 7.4 times as great as use values reported for bottlenose dolphins and sea otters. Pure existence value, without any current or future option to observe the wildlife populations, is 11.6 times as great as use value for the seals, 7.3 times as great for the whales, and 5.5 times as great for the dolphins and sea otters. These results provide evidence that existence values for marine wildlife, where all on-site "use" is precluded, are significantly greater than use values, and they vary for different environmental goods. Even if marine mammals are inaccessible for viewing and impart no regional tourism impacts, societal damages due to marine pollution can still occur. Evidence of the mmeasure of such damages can be found by investigating the existence values which households attach to preservation of marine mammal populations.

Exp. 1: journalistic media exposure to the animal
yes = 1; no = 0
(whales, dolphins, sea otters, elephant seals)

Exp 2: exposure to live animals in captive display
yes = 1; no = 0
(whales, dolphins, sea otters, elephant seals)

Exp 3: exposure to the animals in the wild
yes = 1; no = 0
(whales, dolphins, sea otters, elephant seals)

Exp 4: exposure to the animals in the wild off California's coast in 1983
yes = 1; no = 0
(whales, dolphins, sea otters, elephant seals)

MC: number of miles residing from the California coast

Fam. Size: number of residents in the household

Age: age of respondent

Sex: sex of respondent
female = 1; male = 0

Income: annual household income

Education: respondent's years of education

Hunt/fish: any hunting/fishing done by a household member?
yes = 1; no = 0

Env.Org.: any resident a member of an environmental organization?

AV26-28: Avidity scales for non-fishing and fishing marine recreation
0 = none; 10 = extreme avidity

AV29-31: Avidity scales for species conservation and wilderness preservation
0 = none; 10 = extreme avidity

The response data in Table 8 warrant some discussion on the ability of surveys to elicit information about WTP to protect specific wildlife groups. Familiarity with the animals through communications media (Exp 1) is quite high, near 80% for all groups except the northern elephant seal about which only 60% had previous information. Of interest are the number of yes response to the other three exposure categories. For whales, 30% of the respondents reported seeing gray or blue whales in live captive displays, and 47% reported having seen northern elephant seals. The former is impossible and the latter highly unlikely. Furthermore, while reporting of 1983 on-site sitings of dolphins and sea otter (Exp 4=23% of respondents) is reasonable, some of the individuals who sited whales were those with blue whale questionnaires. Also, 23% reported on-site observation of northern elephant seals. Again, the two latter sitings are highly improbable for anyone but trained biologists. Thus, if we believe respondents are attempting to answer honestly, we might hypothesize that they are able to distinguish by sub-orders, since the pictures and information supplied should allow respondents to group animals into categories such as whales, dolphins/porpoises, sea otters, and seals/sea lions. However, at least some respondents are either unable or unwilling to make the more narrow species distinction (e.g., <u>northern</u> <u>elephant</u> seals) requested in this survey.

survey may be stating some amount which is a base amount for the wildlife category in general. The similarity of stated willingness to pay for each of the four mammal groups lends some suppport for this hypothesis, though it does appear that some additional amount over and above the "base" is added for particular species of marine wildlife. If, for example, we posit that the base amount is, at most, that stated for the least valued species in the survey, the dolphin, then the additional amount stated as a willingness to pay strictly for protection of the California sea otter is the difference -- or $3.02 per household annually (see Table 9). Aggregating over 8.4 million households in California, the annual societal value attached to preservation of the California sea otter is approximately $25.4 million. Again, this estimate does not include values which may exist for non-Californians; in the case of this animal, such values undoubtedly exist for some individuals, as evidenced by a worldwide membership in and contributions to a non-profit organization formed to support actions to protect the sea otter.

Valuing Incremental Changes in Marine Wildlife Populations

A difficult valuation task is that of determining societal losses associated with incremental reductions in wildlife populations in the event of a marine pollution incident. Although this study looked at animal population reductions only in the context of reducing the species' to historical lows before protection programs were enforced, some evidence of individuals' ability to place values on incremental changes (but improvements) in the marine environment is provided in Table 11.

The WTP responses reported in Table 9 were elicited by a hypothetical situation wherein respondents were asked to estimate their willingness-to-pay to avoid a reduction in the mammal populations below current levels without public protection programs. Respondents were also asked to provide estimates of WTP to obtain incremental increases in the levels of current mammal populations. Given that Situation C is defined as the current population level, Situation B is an increment over the current level (between C and A), and Situation A is a final increment up to the historically high level (before hunting by non-natives). The WTP responses to the questions (Question 16: C -> B, Question 17: B -> A) are shown in Table 11. Again, it appears that responses are generally higher for whales and sea otters than for dolphins and elephant seals.

Although the willingness to pay values shown in Table 11 are for increases in the marine mammal populations, it does appear that the respondents exhibit diminishing marginal utility when valuing incremental changes. Furthermore, it may be reasonable to assume that the stated WTP values for increments to the populations are rough estimates for decrements to those animal groups, if the decrements are not large enough to reduce the populations to the endangered levels (as in the values shown in Table 9). Also, future research efforts could incorporate questions on willingness to pay to avoid incremental losses in marine wildlife populations.

REFERENCES

Belsley, D.A., E. Kuh and R.E. Welsh. 1980. Regression Diagnostics John Wiley and Sons, New York, N.Y.

Bishop, R.C. and T.A. Herberlein. 1979. "Measuring Values of Extra-Market Goods: Are Indirect Measures Biased?" American Journal of Agricultural Economics 61(5):926-930.

Boyle, K.J. and R.C. Bishop. 1985. The Total Value of Wildlife Resources: Conceptual and Empirical Issues. AERE Workshop on Recreation Demand Modeling. Boulder, CO.

Brookshire, D.S., L.E. Eubanks and A. Randall. 1983. "Estimating Option Prices and Existence Values for Wildlife Resources," Land Economics 59:1-15.

Burness, H.S., R.G. Cummings, A.F. Mehr and M.S. Walbert. 1983. "Valuing Policies Which Reduce Enviromental Risk," Natural Resources Journal 23(3):675-682.

Carson, R.T. and R.C. Mitchell. "Some Comments on the State of the Arts Assessment of the Contingent Valuation Method Draft Report," Valuing Public Goods: An Assessment of the Contingent Valuation Method (R.G. Cummings, D.S. Brookshire, and W.D. Schulze, Eds.), forthcoming.

Cummings, R.G., D.S. Brookshire and W.D. Schultze (eds). 1984. Valuing Public Goods: A State of the Art Assessment of the Contingent Valuation Method, Forthcoming.

Desvousges, W., V.K. Smith and M. McGivney. 1982. "A Comparison of Alternative Approaches for Estimating Recreation and Related Benefits of Water Quality Improvement," prepared for the U.S. Enviromental Protection Agency, Research Triangle Park, North Carolina.

Dillman, D.A. 1978. Mail Telephone Surveys: The Total Design Method New York: Wiley.

Fisher, A. and R. Raucher. 1984. "Intrinsic Benefits of Improved Water Quality: Conceptual and Empirical Perspectives". Applied Micro-Economics, Vol. 3, eds. V.K. Smith and A.D. Witte. Greenwich, CT: Jai Press.

Goudy, W.J. 1978. "Nonresponse Effects on Relationship Between Variables," Public Opinion Quarterly 40(3):360-369.

Hageman, R.K. 1985. Valuing Marine Mammal Populations: Benefit Valuations in a Multi-Species Ecosystem. National Marine Fisheries Service Admin. Report LJ-85-22. La Jolla, CA.

Huppert, D.D. and C.J. Thomson. 1984. "Demand Analysis of Partyboat Angling in California Using the Travel Cost Method," NMFS Report #LJ-84-06.

Kahneman, D. 1984. "Comments on Valuing Enviromental Goods: A State of the Art Assessment of the Valuation Method," R.G. Cummings, D.S. Brookshire and W.D. Schulze (eds), U.S. Enviromental Protection Agency Conference, Palo Alto, California.

Krutilla, J.V. 1967. "Conservation Reconsidered," American Economic Review 57(4):777-786.

Madalla, G.S. 1977. Econometrics, New York: McGraw-Hill.

Mendelsohn, R.O. 1984. The Benefits of Preserving Endangered Species: With Special Attention to the Humpback Whale. National Marine Fisheries Service Admin. Rep. H-84-14. Honolulu, HI.

McConnell, K.E. and I. Strand. 1981. Measuring the Cost of Time in Recreation Demand Analysis: An Application to Sport Fishing. American Journal of Agricultural Economics, 66(5):153-156.

Miller, J.R. and M.J. Hay. 1984. Estimating Substate Values of Fish and Wildlife. 49th North American Wildlife and Natural Resources Conference, Boston, MA.

Randall, A. and J.R. Stoll. 1983. Existence Valuation in a Total Valuation Framework. In: R. Rowe and L. Chestnut (eds.), Managing Air Quality and Scenic Resources at National Parks and Wilderness Areas, Westview Press: Boulder, Colorado.

Schulze, W.D., R.G. Cummings, D.S. Brookshire, M.A. Thayer, R. Whitworth, and M. Rahmatian. 1983. Experimental Approaches for Valuing Enviromental Commodities, Methods Development in Measuring Benefits-of Enviromental Improvements, U.S. Enviromental Protection Agency.

Wellman, J.D., E.G. Hawk, J.W. Roggenbuck, and G.J. Buhyoff. 1980. Mailed Questionaire Surveys and the Reluctant respondent: An Empirical Examination of Differences Between Early and Late Respondents. Journal of Leisure Research 12(2):164-173.

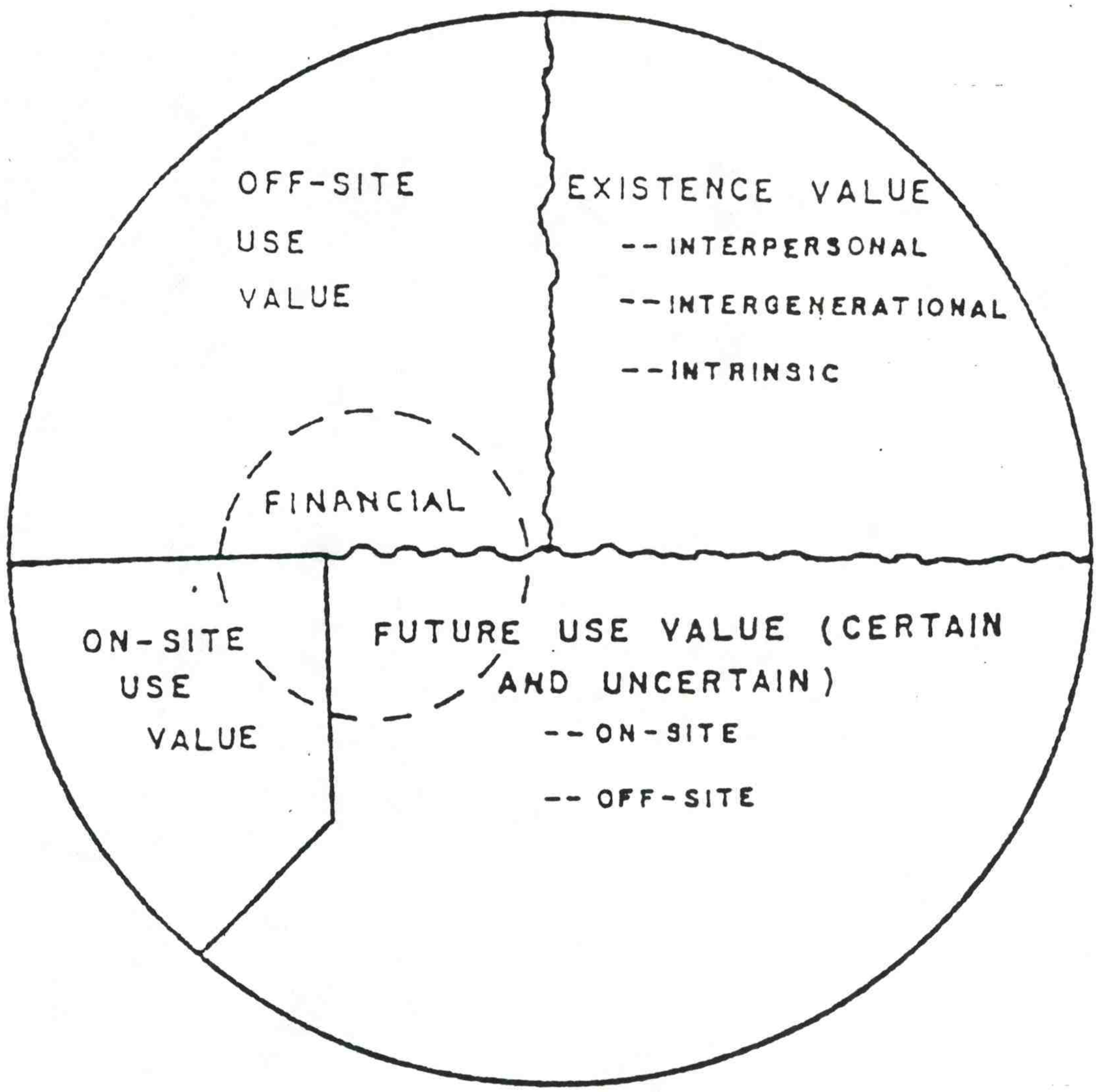

Figure 1. Total value framework.
(from Randall and Stoll, 1983)

Table 1. MEAN WTP/YEAR PER HOUSEHOLD, BY SURVEY GROUP

Gray Whales first
n=93

	Gray Whales	Bottlenose Dolphins	Sea Otters	Elephant Seals
Mean	$26.98	$22.00	$26.12	$21.69
Standard eviation	49.10	43.61	45.38	41.46

Blue Whales first
n=87

	Blue Whales	Bottlenose Dolphins	Sea Otters	Elephant Seals
Mean	$28.78	$23.16	$24.97	$23.13
Standard Deviation	52.56	48.26	48.40	49.06

Table 2. MEAN WTP/YEAR PER HOUSEHOLD, BY RESPONSE GROUP[a]

	Whales Blue and Gray		Bottlenose Dolphins		California Sea Otters		Northern Elephant Seals	
	I	II	I	II	I	II	I	II
Mean	$28.98	$26.15	$25.86	$17.62	$27.63	$22.49	$25.03	$18.46
Standard Deviation	56.58	40.54	56.11	22.43	57.09	24.37	55.82	20.89
t-statistic[b] (degrees of freedom)	.37 (178)		1.17 (173)		.71 (172)		.94 (172)	

[a] Response Set I received after initial mailing and reminder; Response Set II received after final mailing.

[b] The null hypothesis, $mean_I = mean_{II}$, is rejected for $t \geq 2.617$ (99% confidence).

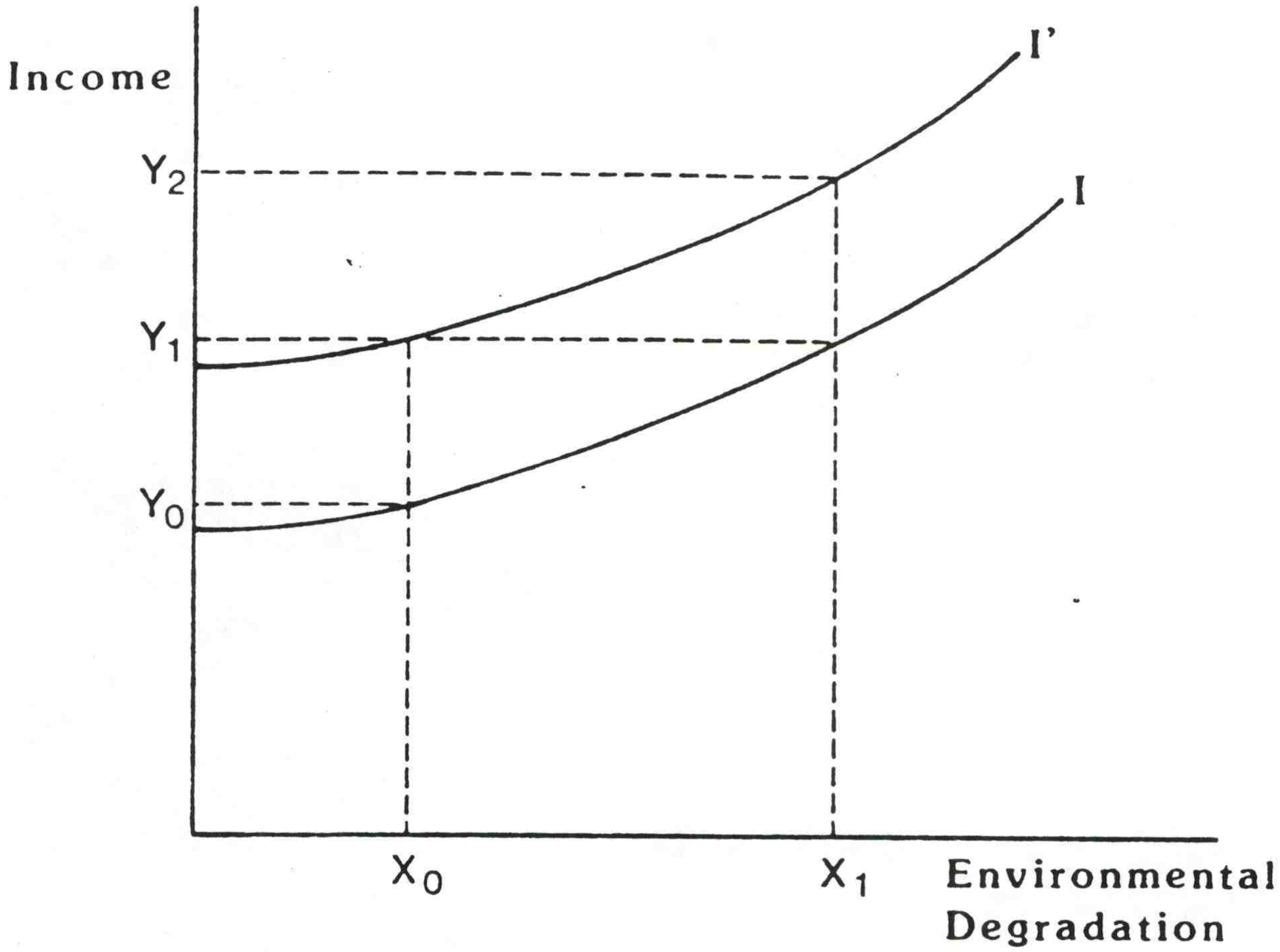

Figure 2. Comparison between willingness-to-pay and willingness-to-accept for changes in environmental quality.

Deterioration: $X_o \longrightarrow X_1$

Maximum WTP = EV_D = $Y_1 - Y_o$
(property rights do not rest with respondent)
Minimum WTA = CV_D = $Y_2 - Y_1$
(property rights rest with respondent)

Improvement: $X_1 \longrightarrow X_o$

Maximum WTP = CV_I = $Y_1 - Y_o$
(property rights do not rest with respondent)
Minimum WTA = EV_I = $Y_2 - Y_1$
(property rights rest with respondent)

Table 3. INITIAL BID[a]

	Whales Gray and Blue	Bottlenosed Dolphins	California Sea Otters	Northern Elephant Seals
Mean	$16.29	$13.90	$15.47	$13.57
Standard Deviation	27.12	25.68	26.69	25.02

[a] Responses to question #9. (See the survey, Appendix to Chapter IV.)

Table 4. ADJUSTMENTS TO BID[a]

	Whales (Gray and Blue)		Bottlenosed Dolphins		California Sea Otters		Northern Elephant Seals	
	#10	#14A	#10	#14A	#10	#14A	#10	#14A
Mean	+$12.34	-$ .77	+$ 9.29	-$ .62	+$ 9.96	+$ .13	+$ 9.23	-$ .42
Standard deviation	27.59	7.22	22.74	6.28	22.67	7.86	22.63	4.61
t statistic[b]	6.0	1.4	5.4	1.3	5.8	.2	5.4	1.2
(degrees of freedom)	(179)		(174)		(173)		(173)	

[a] See questions #10 and #14A in the survey, Appendix to Chapter IV.
[b] The null hypothesis, that the mean value shown is significantly different from zero, is rejected for $t \geq 2.617$ (99% confidence).

Table 5A: CORRELATION MATRIX, WHALES (GRAY AND BLUE)

	WTP	EXP	Miles	Fam.Sz	Age	Sex	Income	Educ.	Hunt/fish	Env.Org.	AVM1
WTP	1.										
EXP	.09	1.									
Miles	-.04	-.10	1.								
Fam. Sz.	-.05	.02	.03	1.							
Age	-.15	.02	.02	-.01	1.						
Sex	.05	-.14	.11	.13	.08	1.					
Income	.09	-.04	-.05	-.43	-.12	-.11	1.				
Education	.12	.03	-.14	.16	-.18	-.19	.22	1.			
Hunt/fish	-.01	-.04	.15	.25	.23	-.01	-.10	-.22	1.		
Env. Org.	.09	.12	.06	-.23	.21	-.01	.03	.21	.02	1.	
AVM1	.16	.13	-.09	-.06	-.31	-.06	.01	.09	.37	.03	1.
AVM2	.24	.20	-.06	.03	.06	.04	.19	-.14	-.05	.33	.36

Table 6. ESTIMATED COEFFICIENTS, OLS REGRESSIONS
DEPENDENT VARIABLE=WTP (Q 11±Q 14A)
(t-values in parentheses).

	Whales (Gray and Blue)		Bottlenose Dolphin		California Sea Otters		Northern Elephant Seals	
	All Data	Without Outliers	All Data	Without Outliers	All Data	Without Outliers	All Data	Without Outliers
Constant	4.89	.98	4.94	-3.22	4.58	-1.28	2.65	4.77
EXP	2.27	.84	4.33	3.65*	4.60*	3.04*	5.72*	1.57*
	(.68)	(.43)	(1.58)	(2.64)	(1.77)	(2.16)	(2.38)	(1.21)
MC	-.003	.001	-.419	-.54	-.05	-.01	.03	.03
	(-.27)	(.16)	(-.38)	(-.98)	(-.46)	(-.20)	(.28)	(.47)
FSZ	-2.96	-1.34	-2.97	-1.20	-4.11*	-2.26*	-2.28	-1.16
	(-1.14)	(-.76)	(-1.25)	(-1.01)	(-1.69)	(-1.72)	(-.97)	(-.92)
Age	-.57*	-.37*	-.45*	-.19*	-.45*	-.21*	-.44*	-.24*
	(-2.33)	(-2.17)	(-1.97)	(-1.71)	(-1.98)	(-1.65)	(-1.94)	(-1.99)
$Y_{10,000}$	3.11*	3.13*	1.65	1.58*	1.78	1.67*	1.42	1.45*
	(1.88)	(2.76)	(1.09)	(2.09)	(1.14)	(1.98)	(.94)	(1.79)
AVM2	5.53*	4.00*	3.86*	2.41*	4.72*	3.36*	3.96*	2.54*
	(3.58)	(3.79)	(2.69)	(3.34)	(3.37)	(4.26)	(2.90)	(3.46)
Degrees of freedom	173	171	168	165	167	164	167	165
R^2	.08	.10	.07	.13	.08	.12	.08	.08

*Significant at the 95% level, t≥1.645.

Table 7. RESPONSES OF LIKELY OUTLIERS

Responses	OUTLIER 1 W	OUTLIER 1 B.D.	OUTLIER 1 S.O.	OUTLIER 1 E.S.	OUTLIER 2 W	OUTLIER 2 B.D.	OUTLIER 2 S.O.	OUTLIER 2 E.S.	OUTLIER 3[a] W	OUTLIER 3[a] B.D.	OUTLIER 3[a] S.O.	OUTLIER 3[a] E.S.
WTP/Year	350,	350,	350,	350	400,	400,	400,	400	150,	150,	150,	100
Exp. 1	yes,	yes,	yes,	yes	yes,	yes,	yes,	yes	yes,	yes,	yes,	yes
Exp. 2	no,	yes,	yes,	yes	yes,	yes,	yes,	yes	yes,	yes,	yes,	yes
Exp. 3	yes,	yes,	yes,	yes	no,	no,	no,	yes	yes,	yes,	yes,	yes
Exp. 4	no,	no,	yes,	no	no,	no,	no,	no	no,	no,	no,	yes
Miles to Coast		15				15				6		
Fam. Size		3				1				3		
Age		32				33				25		
Sex		Female				Male				Female		
Income		$50,000				$15,000				$50,000		
Education		18 Yrs				19 Yrs				12 Yrs		
Hunt/Fish		no				no				yes		
Env. Org.		no				no				no		
AV26		10				10				7		
AV27		1				0				9		
AV28		0				0				3		
AV29		10				10				7		
AV30		10				10				5		
AV31		10				10				5		

[a] An outlier for sea otter and dolphin data sets only.

Table 8. MEAN VALUES: WTP AND SOCIO-ECONOMIC CHARACTERISTICS (OUTLIERS REMOVED)

Variables	Whales	Bottlenose Dolphins	California Sea Otters	Northern Elephant Seals	Standard Deviation (Weighted Average)
WTP/Year	$23.95	$17.73	$20.75	$18.29	$28.39
Exp. 1 (yes=1)	.78	.79	.80	.59	.43
Exp. 2 (yes=1)	.29	.82	.63	.47	.45
Exp. 3 (yes=1)	.35	.47	.53	.31	.49
Exp. 4 (yes=1)	.16	.17	.25	.10	.37
Miles to Coast	22.8	23.0	22.9	22.5	30.61
Family Size	2.67	2.67	2.69	2.68	1.45
Age	42.5	42.3	42.3	42.2	15.1
Sex (0=Male)	.39	.38	.39	.39	.49
Income ($/year)	$35,302	$35,314	$34,994	$35,081	$22,739
Education (years)	15.3	15.4	15.4	15.3	2.87
Hunt/fish (0=no)	.39	.37	.37	.38	.40
Env. Org. (0=no)	.19	.18	.18	.18	.40
AV26 (0-10)	5.5	5.5	5.6	5.6	3.2
AV27 (0-10)	2.6	2.5	2.5	2.6	3.1
AV28 (0-10)	2.0	2.0	2.0	2.0	2.7
AV29 (0-10)	6.9	6.9	6.9	6.9	2.7
AV30 (0-10)	7.3	7.3	7.3	7.2	2.6
AV31 (0-10	7.8	7.8	7.7	2.6	

Table 9. MEAN WTP/YEAR[a] PER HOUSEHOLD, (OUTLIERS REMOVED)[b]

	Whales (Gray and Blue)	Bottlenose Dolphins	California Sea Otters	Northern Elephant Seals
Mean	$23.95 ($27.85)	$17.73 ($22.57)	$20.75 ($25.56)	$18.29 ($22.39)
Standard Deviation	34.82 (50.67)	23.58 (45.80)	25.77 (46.73)	24.19 (45.16)
Number of Observations	178 (180)	172 (175)	171 (174)	172 (174)
t value[c]	9.18	9.83	10.52	9.89
Maximum	$250 ($400)	$135 ($400)	$132 ($400)	$145 ($400)
Minimum	$0 ($0)	$0 ($0)	$0 ($0)	$0 ($0)

[a] Q 11± Q14A
[b] For comparison, values for all respondents, including likely outliers, are shown in parentheses.
[c] The null hyposthesis is WTP>0; reject if t≥2.326 (99% confidence).

Table 10. COMPARISON OF DIFFERENCE IN MEAN VALUES, BY SPECIES, FOR PAIRED RESPONSES

H_o:	$\bar{W}-\bar{D}=0$	$\bar{W}-\bar{SO}=0$	$\bar{W}-\bar{ES}=0$	$\bar{SO}-\bar{D}=0$	$\bar{D}-\bar{ES}=0$	$\bar{SO}-\bar{ES}=0$
t-Statistic: (degrees of freedom=171)	3.529	1.057	2.172	1.601	.070	2.988
Reject; t≥1.645: (95% confidence)	yes	no	yes	~no (very close)	no	yes

Table 11. WTP/YEAR PER HOUSEHOLD, OUTLIERS REMOVED.

	Whales (Gray and Blue)	Bottlenose Dolphins	California Sea Otters	Northern Elephant Seals
C⟶B				
Mean	$ 6.95	$ 4.58	$ 6.12	$ 4.20
Standard Deviation	17.89	12.92	13.83	11.27
B⟶A				
Mean	$ 3.70	$ 2.78	$ 3.55	$ 2.57
Standard Deviation	11.86	11.19	11.40	10.09

Table 12. MEAN WILLINGNESS-TO-PAY RESPONSES:[a]

BREAKDOWNS BY USE AND NON-USE

	Whales (Blue and Gray)	Bottlenose Dolphins	California Sea Otters	Northern Elephant Seals
Non-Consumptive Use	$ 2.34 (9.3%)	$ 2.21 (11.9%)	$ 2.49 (12%)	$ 1.16 (6.2%)
Option Price	$ 5.79 (22.9%)	$ 4.15 (22.4%)	$ 4.71 (22.6%	$ 4.16 (22.1%)
Existence Value	$17.15 (67.9%)	$12.20 (65.9%)	$13.62 (65.4%)	$13.50 (71.7%)

[a] Total WTP, the sum of each column, differs slightly from the values reported in previous tables because a small number of respondents did not break down WTP into these categories. Their valuations were removed from the results shown in this table.

Risk-Sharing and Liability in the Control of Stochastic Externalities

by

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Paper prepared for the Association of Environmental and Resource Economists' Workshop on Marine Pollution and Environmental Damage Assessment, Narragansett, Rhode Island, June 5-6, 1986.

I. Introduction

Many forms of pollution are stochastic in the sense that they result from accidental spills or releases rather than continuous (intentional) emissions. These include the highly publicized problems resulting from spills of hazardous substances during transport by land or sea and the contamination of groundwater supplies by unintentional releases from landfills. Recognition of these forms of stochastic externalities has given rise to questions concerning their control. For example, Just and Zilberman (1979) compare the effects of lump sum taxes and subsidizes on a firm's incentive to undertake safety. The control of oil spills through the use of liability rules has been studied by Conrad (1980) and by Opaluch and Grigalunas (1984). A more general treatment of the control of accidents appears in a series of papers by Shavell (1980, 1982, 1984a, 1984b).

The choice of any policy for controlling stochastic externalities generally has two effects: an incentive effect and a risk sharing effect. The incentive effect provides the impetus for firms to take actions to increase safety and thus reduce the probability of accidents. The risk sharing effect stems from the fact that the policy choice dictates an allocation of risk and the amount of risk that parties must bear can have welfare effects. Although the incentive effects of alternative policies have been well-recognized, in general the risk sharing effects have been ignored. (An exception is Shavell (1982).) However, recent liability cases and their ripple effects suggest that risk may be a very important consideration in decisions regarding activities that could impose substantial externalities.[1/] Thus, the allocation of risk under alternative policies would appear to be an important factor in the choice of a control policy.

The purpose of this paper is to analyze alternative policies for controlling stochastic externalities in terms of both their incentive and their risk-sharing effects. The policies that are considered depend upon whether the actions of the polluters that affect the probability of a given pollution event are observable or not. When actions are observable, regulation of those actions is possible and the policies considered include regulation and full liability (i.e. ex post liability for the full amount of damages). However, when actions are unobservable, regulation is not possible. Instead liability rules can be used to induce safety, and we consider alternative rules including zero, partial and full ex post liability. The paradigm that provides the basis for the analysis is the principal-agent model that is popular in studies of sharecropping, alternative wage contracts and the organizational structure of firms. The relevance of this model to problems of environmental externalities was first noted by Shavell (1979a).

The paper is organized as follows. In the next section, the pure risk sharing problem (without incentives effects) is presented to provide an understanding of the role of liability rules in the Pareto efficient allocation of risk. The following section presents the model for the case where actions are unobservable and thus incentives for safety must be provided. This section highlights the basic tradeoff between risk sharing and incentives when polluters are risk averse. Section IV compares the use of regulation and ex post liability when the polluter's actions are observable (and can thus be regulated). An interim summary of the results is then presented, followed by a discussion in Section VI of how those results and conclusions would change if insurance were available to spread risk. The final two sections discuss some limitations of the analysis that suggest

directions for further research and implications of the analysis for the control of stochastic marine pollution.

II. Liability and Risk-Sharing Without Incentive Problems

Embodied in many federal statutes is an attempt to control environmental pollution through imposing strict liability for damages on the responsible parties [Opaluch, (1984)]. Although there are often limits on the nature and the extent of the liability,[2/] the basic philosophy is that those responsible for the activity that is causing an environmental problem should pay the costs of clean-up or compensation. The use of this approach was intended to encourage all parties involved in the generation, transportation and disposal of polluting substances to take steps to minimize the possibility of environmental damage from their use. In addition, strict liability is often imposed through state or federal courts under the law of torts.

Although strict liability can be used as a way of internalizing the pollution externality,[3/] it also has important implications for the allocation of risk. For example, since future clean-up costs or damages are uncertain and thus are viewed from the present as a random variable, a strict liability rule places all of the risk associated with the level of future costs on the responsible parties. Although this has advantages in terms of providing proper prevention incentives (see Section III), it is not necessarily an optimal allocation of the risk. We consider below a conceptual framework for analyzing optimal risk-sharing in the context of stochastic pollution.

The economic assessment of optimal risk-sharing is usually considered within the context of the more general problem of risk-sharing and incentives. This broader problem, embodied in what has generally been

referred to as the principal-agent problem, has been studied by many authors, including Stiglitz (1974), Ross (1973), Shavell (1979a), Holmstrom (1979), and Grossman and Hart (1983). The analysis is generally in the context of sharecropping, labor contracts, insurance contracts, or the organizational structure of firms. In addition, Shavell (1979a) discusses its applicability to a comparison of strict liability and negligence standards in controlling stochastic externalities such as oil spills. Leland (1978) has used a similar model to analyze OCS leasing policies. Our purpose here is to consider the application of the general framework used in this model to the question of risk-sharing for stochastic pollution.

In this context, the model takes the following form. Let c denote the value of clean-up costs or damages[4/] associated with a future pollution event. From the present perspective, c is viewed as a random variable since future clean-up costs are not known. In this discussion of optimal risk sharing, we assume that the distribution of c is not affected by the actions of the polluter. This assumption is relaxed in the following section where the problem of incentives is considered. Let f(c) be the amount paid by the polluter[5/] for clean-up. Thus, the costs that must be borne by the victim (or by the public sector) are equal to c-f(c). Note that if f(c) is constant, i.e. independent of c, then in a legal sense polluters have no ex post "liability" since the amount they pay does not depend on the damages actually incurred. The payment scheme is instead analogous to an ex ante payment to a trust fund to be used for clean-up. Alternatively, if $f'(c)\neq 0$ then polluters are subject to at least some ex post liability, with $f'(c)=1$ implying full liability.[6/]

Let $V(v_o-c+f(c))$ represent the victim's utility function and let $U(u_o-f(c))$ represent the polluter's utility function, where v_o and u_o are the

initial wealth levels of the victim and the polluter respectively. Then Pareto optimal risk-sharing is given by a liability rule f that satisfies

$$\max EV(v_o - c + f(c)) \text{ subj. to } EU(u_o - f(c)) \geq \bar{U} \qquad (1)$$

where $\bar{U}$ is the polluter's reservation level of utility. The first order conditions for the optimal f require that

$$V'(v_o - c + f(c)) = \lambda U'(u_o - f(c)) \qquad (2)$$

where $\lambda \geq 0$ is the Lagrangian multiplier on the constraint in (1).[7/] Equation (2) defines the optimal level of risk-sharing between the victim and the polluter if the associated second order conditions are met.

To see the implications of (2) for the first best liability rule, differentiate (2) with respect c and solve for f'(c) to get

$$f'(c) = \frac{V''}{V'' + \lambda U''}. \qquad (3)$$

This highlights the importance of the second derivatives of the utility functions, which reflect the attitudes toward risk. For example, consider the implications of (3) in the context of the following alternative cases.

Case 1: Risk Averse Victim ($V''<0$), Risk Neutral Polluter ($U''=0$). In this case, (3) implies that $f'(c) = 1$, and thus $f(c) = c+k$ for some constant k. As noted above, this corresponds to a full liability rule since any increases in clean-up costs are borne fully by the polluter. Thus, when the polluter is risk neutral but the victim is not (so that risk is costly to the victim but not to polluters), then full liability results in optimal risk-sharing with the polluter bearing the full risk associated with future clean-up costs.

Case 2: Risk Neutral Victim ($V''=0$), Risk Averse Polluter ($U''<0$). In this case, (3) implies that $f'(c) = 0$ so that $f(c) = k$. In other words, it is optimal in terms of risk-sharing for the polluter to pay a constant amount

that is independent of the actual realized clean-up costs. Under this rule, polluters have no ex post liability. They are shielded from the uncertainty associated with future clean-up costs and all of the risk is borne by the victim. This is optimal in this case because risk represents a cost to risk-averse polluters but not to a risk neutral victim.

Case 3: Both Victim and Polluter are Risk Averse ($V''<0$, $U''<0$). In this case, $0 < f'(c) < 1$. This implies that $f(c) \neq c+k$ and $f(c) \neq k$. In other words, neither the polluter nor the victim bears the full risk. Instead, risk is shared between them. The polluter is liable for some portion of realized costs, but he is not fully responsible for incremental changes in c. Of course, the optimal allocation of the risk between the two risk averse parties will depend upon the relative magnitudes of U" and V". In the special case where both U and V are quadratic, a fixed apportionment scheme ($f'(c) = \alpha$ for some constant α) is efficient, i.e. each party's ex post payment should be a fixed proportion of the damages regardless of the level of those damages. Under more general utility functions, however, the efficient apportionment will depend on the level of damages.

Case 4: Both the Victim and Polluter are Risk Neutral ($V''=U''=0$). In this case, a unique optimal risk-sharing rule does not exist. Since risk does not represent a cost to either the polluter or the victim, the allocation of risk does not have any welfare effects.

In summary, the pareto optimal rule for allocating risk between the victim and the polluter (in the absence of incentive problems) depends upon their risk attitudes. A full liability rule will yield optimal risk-sharing if polluters are risk neutral but not if they are risk averse. Polluters might be expected to be risk neutral if the magnitude of the possible damages is small relative to the operations of the firm. However, if the potential

damages are large, firms might be expected to be risk averse, a conjecture that is supported by observed purchases of liability insurance. Risk aversion by polluters implies that, ceteris paribus, the victim should bear some (if he is also risk averse) or all (if he is risk neutral) of the risk associated with the level of future clean-up costs.

In many pollution cases, the "victim" can be best thought of as the public sector since public funds are used for clean-up. The question of risk aversion by the public sector has been addressed by Arrow and Lind (1970), who argue that public decisions should not reflect risk aversion if the risks can be spread sufficiently. In our context, this would suggest that, if the public's share of clean-up costs, c-f(c), is spread across many taxpayers, then the public should perhaps be viewed as risk neutral. However, if public funds are insufficient so that some part of the public's share of the damages are borne in the form of residual pollution, then these costs are borne not by the taxpayers in general but instead by those who live in the vicinity of the sites that receive less than full clean-up. In this case, the costs are not spread. Thus, public sector decisions should reflect private (i.e. victim) attitudes toward risk [Fisher (1973)] and under optimal risk-sharing polluters would bear some (or all) of the risk.

III. Liability and Risk-Sharing with Incentive Problems

The above analysis assumes that the distribution of the random variable representing damages (i.e. clean-up costs) is not affected by the actions of polluters, and thus the only issue of concern is the allocation of risk. In reality, the behavior of polluters can often affect the probability of a given magnitude of damages. The appropriate policy response to this effect depends on whether the preventive actions are observable by the regulatory

agency. If those actions are observable and can be monitored (and thus non-compliance detected), then direct regulation is possible. When preventive care cannot be easily monitored, then direct regulation is not possible, but an indirect incentive mechanism such as a liability rule can be used to induce a certain level of abatement or care. In this section, the above analysis is modified to include the need to induce a certain behavior when direct monitoring is not possible. In the following section, the possibility of using direct regulation is considered.

Let a be the level of safety or preventive care taken by the polluter and let g(c,a) be the density function of clean-up costs (damages) given a. In this case, the Pareto efficient payment scheme is given by the solution to

$$\max_{\{a,f(\cdot)\}} \; EV(v_o - c + f(c)) \tag{4a}$$

$$\text{subject to} \quad EU(u_o - a - f(c)) \geq \bar{U} \tag{4b}$$

$$EU_a = 0 \tag{4c}$$

where the second constraint states that the polluter chooses the level of a that maximizes his expected utility.[8/] This constraint reflects the need to motivate the polluter to undertake abatement. This problem can be written more explicitly as

$$\max_{\{a,f\}} \; V(v_o - c + f(c))g(c,a)dc \tag{5a}$$

$$\text{subject to} \quad U(u_o - a - f(c))g(c,a)dc \geq \bar{U} \tag{5b}$$

$$[U(u_o - a - f(c))g_a - U'(u_o - a - f(c))g(c,a)]dc = 0 \tag{5c}$$

where $g_a = \partial g/\partial a$. The optimal fee schedule must then satisfy the following condition:

$$V' = \lambda U' + \mu[(g_a/g)U' - U''] \tag{6}$$

where μ is the multiplier on the second constraint. As long as $\mu \neq 0$, i.e. there is a need to provide an incentive, this condition differs from the condition for optimal risk sharing given in (3). In other words, because of the need to motivate polluters indirectly to take care when their actions are not observable, in general the optimal fee schedule will differ from the one that would generate optimal risk sharing. This makes intuitive sense when one thinks, for example, about the special case of risk averse polluters and risk neutral victims. In this case, optimal risk sharing would imply that victims bear all of the risk, i.e. that the polluters not be subject to any ex post liability. However, in the absence of any liability (or enforceable regulations), polluters have no incentive to be cautious. Thus, there is a trade-off between risk sharing and incentives; greater liability implies greater behavioral incentives but also greater risk for polluters.

Of course, this trade-off disappears in special cases. For example, when polluters are risk neutral, i.e. $U''=0$, one can show that $\mu=0$ and thus the problem reduces to one of just optimal risk sharing. In this case, full liability is optimal, since it provides the correct incentive for precaution while placing all of the risk on the risk neutral party. Likewise, when $g_a=0$, i.e. when polluter's actions do not influence the distribution of c, then again the problem reduces to optimal risk sharing since polluters will choose a zero level of precaution regardless of the fee schedule.

In general, however, neither full liability nor a fixed payment that is independent of actual damages is optimal when the dual goals of risk-sharing and incentives are considered. Instead, as long as polluters are risk averse, some form of partial liability is preferred. The extent of that partial liability would depend on the extent of the polluter's and victim's risk aversion and the strength of the incentive effect.

IV. Ex Ante Regulation vs. Ex Post Full Liability

In the previous section, the assumption that the actions of polluters were unobservable implied that direct regulation was not possible, i.e. the fee paid by polluters could not be a function of a. However, if a is observable, then direct regulation is possible. In particular, a fee schedule of the form

$$f(c) = \begin{cases} 0 \text{ if } a \geq a^* \\ \infty \text{ if } a < a^* \end{cases} \qquad (7)$$

would be equivalent to requiring the firm to abate to a^* and then absolving the firm of any liability for actual damages incurred.[9] Thus, the basic framework outlined in the previous section can also be used to compare the use of ex ante regulation and ex post liability in controlling stochastic externalities.

Previous comparisons of these two approaches [e.g. Shavell (1984a, 1984b), White and Wittman (1983), Johnson, Kolstad and Ulen (1985)] have assumed that all parties are risk neutral. They thus focus on the incentive effects of the two alternatives and ignore the risk sharing effects. In this case the relative desirability of the two approaches depends upon the assumptions that are made about system imperfections. For example, Shavell (1984a) argues that the ability of polluters to escape successful suits or avoid full payment for damages because of asset availability tend to make regulation more desirable, while the inability of regulators to distinguish ex ante among firms threatening different levels of harm tends to favor the use of ex post liability rules. Johnson, Kolstad and Ulen (1985) consider the impact of evidentiary uncertainty, i.e. uncertainty about the legal standard that a potential polluter would be held to in court. In these

cases, the inability of the regulation and liability approaches to ensure an efficient level of precaution stems from some assumed imperfection in the regulatory or legal system. In the absence of these imperfections, the two alternatives would be equally efficient. This result does not hold, however, when the assumption of risk neutrality is relaxed because the allocation of risk under the two approaches becomes a factor in determining their relative desirability. In this section, we demonstrate that even in the absence of system imperfections the two approaches are not equally desirable when risk aversion is allowed.

To simplify the analysis (and make it more comparable with previous work), assume that a pollution event (i.e. a spill or release) either occurs or does not occur and that if it occurs the damages are equal to d. Assume that the actions of the polluter affect the probability that an accident will occur. Thus, the density function g(c,a) takes the form

$$c = \begin{cases} 0 \text{ with probability } 1-p(a) \\ d \text{ with probability } p(a), \end{cases} \tag{8}$$

where p(a) is the probability of an accident occurring given a precaution or safety level a.

Intuitively, the importance of risk aversion in determining the relative desirability of regulation and full liability can be seen by recalling the well-known fact that risk averse polluters should be willing to pay a premium to eliminate risk. Thus, if compliance with regulations would absolve them of ex post liability, risk averse polluters would be willing to be subjected to a regulatory standard that is more stringent than the level of care they would choose voluntarily under a full liability system. More specifically, let $\tilde{a}$ be the level of precaution that maximizes the polluter's expected

utility under full liability, i.e. $\tilde{a}$ maximize $EU(a) = (1-p(a))U(u_o-a) + p(a)U(u_o-a-d)$. Let $EU(\tilde{a}) = \tilde{U}$ be the firm's maximum expected utility under full liability. The firm should then be indifferent between a full liability system and a policy that couples ex ante payments to victims equal to the expected value of damages under full liability with a regulatory standard of $\tilde{s}$ defined by $\tilde{U} = U(u_o-p(\tilde{a})d-\tilde{s})$ where compliance with the regulation is a sufficient defense against liability. However, if the firm is risk averse then $\tilde{s}>\tilde{a}$. The difference $\tilde{s}-\tilde{a}$ is the risk premium the polluter is willing to pay to get rid of the risk borne under full liability. Thus, under regulation the victims can get more prevention for the same "price" (in terms of the polluter's expected utility).[10/] However, by choosing regulation over full liability they are also subject to more risk. This risk is costless if they are risk neutral but not if they are risk averse. Thus, which of the two alternatives is preferred depends on how the victim trades off increased risk against increased protection.

To see this more explicitly, consider the Pareto efficient regulatory standard, $\bar{a}$. This is given by the solution to

$$\max_{a,k} \quad EV = (1-p(a))V(v_o+k) + p(a)V(v_o+k-d) \tag{9a}$$

$$\text{subject to} \quad U(u_o-a-k) \geq \bar{U} \tag{9b}$$

where k is a lump sum transfer from polluters to victims. (If the efficient transfer $\bar{k}$ were negative, the transfer would be from victims to polluters.) This transfer represents an ex ante, i.e. state-independent, compensation or indemnification that keeps the polluter's utility level at $\bar{U}$. Note that, if polluters are absolved of tort liability by compliance with the standard,[11/] then under the regulatory approach they bear no risk. All of the risk is borne by the victim. In this sense, regulation is equivalent to a fixed ex

ante scheme where the fee is $\bar{a}+\bar{k}$.

Under the alternative policy of imposing full liability without any regulation, firms would be free to choose their level of precaution. Thus, the level of precaution under liability solves

$$\max_{a} \quad (1-p(a))U(u_o-k-a) + p(a)U(u_o-k-a-d). \tag{10}$$

Note that the solution is a function of k, which we denote $a^*(k)$. The level of k necessary to keep the polluter's expected utility at U is then implicitly defined by

$$EU = (1-p(a(k)))U(u_o-k-a(k)) + p(a(k))U(u_o-k-a(k)-d) \equiv \bar{U}. \tag{11}$$

We denote this solution k^* and the corresponding level of precaution $a^* = a^*(k^*)$.

Since the expected utility of polluters has been held at $\bar{U}$ under both policies, we can compare the desirability of the two by comparing the expected utility of victims. Let $\bar{V}$ be the victim's expected utility under the efficient regulatory standard, i.e.

$$\bar{V} = (1-p(\bar{a}))V(v_o+\bar{k}) + p(\bar{a})V(v_o+\bar{k}-d), \tag{12}$$

and let V^* be the victim's expected utility under full liability, i.e.

$$V^* = V(v_o+k^*). \tag{13}$$

Then regulation is preferred to full liability if $\bar{V}>V^*$, and vice versa.

Result:

(a) If victims are risk neutral and firms are risk averse, then $\bar{V}>V^*$, i.e. victims are better off under regulation than under a system of full liability;[12/] (b) If polluters are risk neutral and victims are risk averse, then $V^*>\bar{V}$, i.e. full liability is preferred; and (c) Risk neutrality for both parties implies that the two approaches are equally efficient, i.e. $V^*=\bar{V}$.

These results are consistent with what would be expected from the discussion in the previous sections, and they are proven in the appendix. However, when we move away from these special cases and allow both parties to be risk averse, then unambiguous statements about which policy approach is preferred can no longer be made even in the absence of system imperfections. The analysis suggests that something in between full liability and the sole use of regulation i.e. some form of partial liability plus regulation, would be efficient.

V. Summary So Far

If system imperfections (such as evidentiary uncertainty, the difficulty of proving responsibility and limitations on recoverable amounts) are ignored, then the above analysis suggests the following conclusions:

Case 1: Polluters are risk neutral. In this case, both in terms of efficiency and risk sharing, a system of full liability is efficient. Although polluters may be expected to be risk neutral with regard to small risks, the recent furor over the shrinking of the pollution liability insurance market suggests that for large environmental risks firms are not likely to exhibit risk neutrality.

Case 2: Polluters are risk averse, victims are risk neutral and all of the polluter's precautionary actions are observable. In this case regulation alone would be efficient. The allocation of risk would be efficient and the correct incentives could be maintained by setting and enforcing an appropriate regulatory standard. However, in reality, it is unlikely that victims of environmental damages will be risk neutral since the losses can be large relative to an individual's income and, even if government compensation for monetary damages is available, there are likely to be non-monetary

damages that prevent full compensation. In addition, it is unlikely that all of the polluter's actions that influence the probability of a given magnitude of damages will be able to be controlled through regulations. Even if firms have safety procedures or equipment designed to reduce accidents, the care with which these procedures are followed or the equipment maintained is in general not easily (or cheaply) monitored by the regulatory agency.

Case 3: Polluters are risk averse, victims are risk neutral and some of the polluter's precautionary actions are not observable. In this case, sole reliance on regulation is not efficient (even though it optimally allocates risk) because it does not provide the correct incentive to undertake unobservable precautionary actions. Instead, it would appear to be preferable to use a system that couples regulation with a liability rule under which polluters are liable for something less than the full amount of damages. Although the use of ex post liability violates optimal risk-sharing, it is necessary to provide some incentive. In general a system of fixed apportionment, i.e. liability for a fixed proportion of damages regardless of their magnitude, is not efficient, although it might be a reasonable approximation to use in practice.

Case 4: Both polluters and victims are risk averse and all of the polluter's precautionary actions are observable. The efficient policy in this case would be similar to that for Case 3, but for different reasons. In general, some combination of regulation and liability would appear to be efficient, where the liability is for an amount less than the full damages. Here the use of liability is not necessary for incentive purposes but rather to reallocate risk, i.e. provide some form of ensured compensation for risk averse victims. The suggested compensation is not full, however, because full compensation would leave risk averse polluters bearing too much risk.

Case 5: Both polluters and victims are risk averse and some of the polluter's actions are not observable. Again, some form of less than full liability would appear appropriate. If some of the polluter's actions are observable, then coupling the liability system with a regulatory program for those actions would improve the incentive effects without altering its risk-sharing features. This case is perhaps the most likely case for large environmental externalities. It suggests that the joint use of regulation and liability to control stochastic pollution events, such as the combination of RCRA and CERCLA to address hazardous waste dangers, is not necessarily redundant. However, the full liability for clean-up costs imposed by CERCLA may place an inefficient amount of risk on polluters if firms are also held liable for the full amount of third party damages under common law.

In each of the above cases, the regulatory or liability policy would in general require a lump-sum, i.e. state independent, transfer between victims and polluters in order to maintain an acceptable level of expected utility for one of the parties. (This transfer has been denoted k.) It is an ex ante payment of compensation. When the payment is from polluters to victims (as might be expected under the sole use of regulation or less than full liability), it represents ex ante payment for imposing environmental risks and could take the form of fixed payments to a fund such as Superfund to be used for clean-up of existing problems. Alternatively, when the transfer is from victims to polluters (as might be expected under full liability), it represents ex ante payment for imposing financial risks on firms and could take the form of cost sharing or tax breaks to reduce the financial burden associated with undertaking substantial precautionary actions.

VI. The Role of Insurance

The analysis in the previous sections implicitly assumes that risk averse parties are unable to transfer risk through the purchase of first party or liability insurance. In this section we discuss how the existence of insurance markets to spread risk would affect those conclusions. The existence of such markets should not, however, be taken for granted even (or perhaps especially) when risks are very large. For example, in theory under policies that impose risks on polluters (i.e. when $f'(c) \neq 0$), we would expect liability insurance to be available since risk averse polluters would generate a demand for it. Recently, however, the market for liability insurance has nearly collapsed. Thus, although historically they have been able to do so, risk averse polluters may no longer be able to purchase insurance to transfer all liability risks, especially those associated with low probability, high consequence (LP-HC) events. Since the availability of insurance affects the allocation of risk under the alternative policies, whether or not it exists is an important factor in analyzing those policies when polluters or victims are risk averse.

The impact of insurance on the efficiency of different liability rules has been studied by Shavell (1982). The discussion here draws on some of Shavell's results and the well-known fact that a risk averse party can improve his welfare by purchasing actuarially fair insurance. We consider in turn the cases summarized in the previous section.

Case 1: Polluters are risk neutral. In this case, the possibility of purchasing liability insurance is irrelevant since risk neutral polluters would have no incentive to purchase actuarially fair insurance. Full liability is still the efficient approach in terms of both incentives and

risk-sharing. Since victims bear no risk under full liability, their ability to purchase first party insurance is also irrelevant.

Case 2: Polluters are risk averse, victims are risk neutral and the polluter's actions are observable. Again, the availability of insurance does not change the previous conclusions. Regulation alone is still efficient. Victims have no interest in insurance because they are risk neutral and polluters will not purchase any insurance because they do not bear any risk under regulation.

Case 3: Polluters are risk averse, victims are risk neutral and some polluter actions are unobservable. The fact that some actions are unobservable implies that liability insurers will be unable to base premiums on the level of preventive action and as a result moral hazard will exist. Risk averse polluters will purchase less than full coverage for the risks they must bear (Shavell, 1979b). Shavell (1982) has shown that in this case the efficient liability rule is to either (1) impose full liability on the polluters or (2) prohibit liability insurance and impose an appropriate level of partial liability. The net effect of these two alternatives is the same, since under the first one polluters would purchase less than full coverage, leaving them with the same incentives and risks as under the second option. (The banning of liability insurance under the second option ensures that firms cannot further dilute the incentive effects of the partial liability through the purchase of insurance.) In addition, because victims are risk neutral, they are unaffected by the greater risk they bear under the second option.

Case 4: Both polluters and victims are risk averse and the polluter's actions are observable. It is in this case that the availability of insurance has the greatest potential for improving the outcome since the

problem here is only a problem of risk sharing and not a problem of incentives. In the absence of insurance, efficiency requires that risk be divided between polluters and victims. However, if both first party and liability insurance are available, then both victims and polluters can eliminate their risk through the purchase of insurance and the allocation of risk becomes irrelevant. If, on the other hand, only liability insurance is available, then the use of full liability is efficient. Polluters will purchase actuarially fair insurance to transfer the risks they bear under full liability, but, since their actions are observable, insurers will base their premiums on those actions and thus firms will still face the proper incentives. Alternatively, if only first party insurance is available, then the use of a regulatory approach without liability shelters polluters from risk and victims can also avoid risk through the purchase of insurance. Thus, in this case, the choice of an efficient policy depends crucially on the availability of insurance.

Case 5: Both polluters and victims are risk averse and some polluter actions are not observable. Here the incentive problems are the same as in Case 3 because of moral hazard. If liability insurance is available, then the proposed policies are the same as well, provided that the risk averse victims can purchase first party insurance to cover the risks they would bear under the second option where liability insurance is banned with polluters subject to only partial liability. If first party insurance is not available, then the use of full liability is the preferred option since the existence of liability insurance protects risk averse victims from risk.

VII. Conclusions and Limitations

Recent events regarding legal liability for damages due to stochastic pollution and the associated "insurance crisis" suggest that potential polluters exhibit risk aversion with respect to the uncertainty associated with damages. Furthermore, it seems likely that in most cases of stochastic pollution some of the actions of the potential polluter that affect the density function of damages are not easily subject to regulation. When these two conditions exist, the above analysis suggests the following conclusion: (1) If liability insurance is not available to transfer risks, then an efficient policy for the control of stochastic externalities would include the use of both regulation of observable actions and ex post liability, where the liability would be for an amount less than the full amount of damages; and (2) If liability insurance is available, then the use of full liability is efficient since risk sharing can be achieved through the purchase of insurance.

There are several caveats to this conclusion that reflect the limitations of the above analysis. First, as noted previously, imperfections in the regulatory and liability systems have not been included in the model. However, the results of Shavell (1984a) and Johnson, Kolstad and Ulen (1985) suggest that inclusion of system imperfections would not necessarily change the conclusion that the joint use of regulation and some ex post liability is desired.

Secondly, the model of decision making under uncertainty that is used here (the expected utility model) has been the subject of considerable criticism.[13/] The use of an alternative paradigm could lead to different conclusions since the implied perception of risk would be different.

Thirdly, the model used is a short run model that does not capture non-marginal adjustments by polluters or victims. To the extent that the different policies imply different expected costs for either polluters or victims, in the long run they would be expected to respond accordingly. For example, high expected costs under a full liability policy might cause individual firms to leave the industry. Alternatively, the prospect of large uncompensated damages might cause victims to relocate to areas of lower risk. These non-marginal behavioral responses would have implications for the long run effect of any policy choice.

Finally, the static nature of the model,[14/] the omission of administrative and legal costs, and the difficulty of empirically determining the risk aversion characteristics of polluters and victims should be kept in mind when interpreting the conclusions of the analysis.

VIII. Implications for Control of Marine Pollution

The conclusions from the above discussion have implications for the efficient control of marine pollution. The current approach to controlling stochastic forms of marine pollution employs a combination of regulation and ex post liability for damages due to releases of polluting substances. For example, regulations governing the use of the marine environment for transportation and waste disposal have been promulgated pursuant to a number of federal statutes, including the Ports and Waterways Safety Act, the Clean Water Act, the Coastal Zone Management Act, the Deepwater Port Act, the Marine Protection, Research and Sanctuaries Act, the Outer Continental Shelf Lands Act, and the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA). Several of these (Clean Water Act, Deepwater Port Act, and CERCLA) explicitly impose liability on vessel or facility owners or

operators for clean-up costs and damages to natural resources that result from unauthorized discharges into the marine environment. However, in the absence of gross negligence or willful misconduct, the total amount of the liability is limited to a specified dollar amount.

The results discussed above suggest that this joint use of regulation and liability may be justified if the owners/operators of polluting vessels or facilities can undertake actions that affect the probability of a release occurring but are not easily observable (and thus subject to regulation). In addition, the dollar limits placed on their liability can be viewed as a means of sheltering risk averse polluters from some risk, again a goal that is consistent with the above results. However, limiting risk by putting a dollar cap on liability is not generally an efficient means of risk allocation.

In essence, the use of a liability cap implies a system of full liability for small damages and partial liability for large damages. Full liability for small damages is efficient in terms of both risk sharing and incentives if polluters can be considered risk neutral with respect to small damages. As noted above, partial liability for large damages is also efficient if polluters are risk averse with respect to large damages. However, implementing partial liability through a liability cap implies that marginal liability is zero beyond the amount of the cap, i.e. $f'(c) = 0$ for all c in excess of the cap. Since this violates Equation (6), it is not an efficient way to balance risk-sharing and incentive needs. The analysis suggests that a preferred approach would be to hold polluters liable for some percentage of damages once they exceed a certain level. Although theoretically that percentage should not be independent of the magnitude of damages (unless both U and V are quadratic), a constant percentage might be

desirable in practice because of its simplicity and the difficulty of determining empirically the precise form of the optimal nonlinear liability rule. Note, however, that if the polluter becomes very risk averse as damages become very large, then the risk sharing effects would come to dominate the incentive effects and the efficient level of marginal liability would approach zero as damages increased. This would imply that at some point a cap on liability might be desirable.

APPENDIX

1. Claim: If $V''=0$ and $U''<0$ then $\bar{V}>V_*$.

Proof: If $V''=0$, then $V'=\gamma$ for some constant γ and $\bar{V} = \gamma\cdot(v_o+\bar{k}-p(\bar{a})d)$ under regulation. Thus, since $V^*=\gamma(v_o+k^*)$, it is sufficient to show that $k^*<\bar{k}-p(\bar{a})d$.

When $V'=\gamma$, the first order conditions for (9) imply that $p'(\bar{a})d+1=0$ and thus that $\bar{a}$ minimizes $a+p(a)d$. This implies that

$$\bar{a} + p(\bar{a})d \le a^* + p(a^*)d$$

$$p(\bar{a})d \le (a^*-\bar{a}) + p(a^*)d,$$

$$\bar{k}-p(\bar{a})d \ge \bar{k}-(a^*-\bar{a}) - p(a^*)d. \qquad (A1)$$

Furthermore, by concavity of U,

$$U[p(x^*)(u_o-k^*-a^*-d) + (1-p(a^*))(u_o-k^*-a^*)]$$

$$> (1-p(a^*))U(u_o-k^*-a^*) + p(a^*)U(u_o-k^*-a^*-d). \qquad (A2)$$

Finally,

$$(1-p(a^*))U(u_o-k^*-a^*) + p(a^*)U(u_o-k^*-a^*-d) = U(u_o-\bar{k}-\bar{a}) \qquad (A3)$$

since both are equal to $\bar{U}$ by (9b) and (11).

Combining (A2) and (A3) yields

$$U[u_o-k^*-a^*-p(a^*)d] > U(u_o-\bar{k}-\bar{a}),$$

and thus, assuming U is monotonic,

$$u_o-k^*-a^*-p(a^*)d > u_o-\bar{k}-\bar{a}.$$

$$k^* \le \bar{k}-(a^*-\bar{a})-p(a^*)d < \bar{k}-p(\bar{a})d$$

where the last inequality follows from (A1). Q.E.D.

2. Claim: If $U''=0$ and $V''<0$, then $V^*>\bar{V}$.

Proof: If $U''=0$, then $U'=\beta$ for some constant β and by (9b) and (11)

$$\beta(u_o-\bar{k}-\bar{a}) = \beta\cdot(u_o-k^*-a^*-p(a^*)d)$$

$$\bar{k}+\bar{a} = k^*+a^* + p(a^*)d.$$

$$k^* = \bar{k} - (a^*-\bar{a})-p(a^*)d. \qquad \text{(A4)}$$

Furthermore, by concavity of V,

$$V[(1-p(\bar{a}))(v_o+\bar{k}) + p(\bar{a})(v_o+\bar{k}-d)]$$

$$> (1-p(\bar{a}))V(v_o+\bar{k}) + p(\bar{a})V(v_o+\bar{k}-d) = \bar{V}$$

$$V[v_o+\bar{k}-p(\bar{a})d] > \bar{V}. \qquad \text{(A5)}$$

Finally, when $U'=\beta$ then from (10) a^* minimizes $a+p(a)d$

$$a^*+p(a^*)d \le \bar{a}+p(\bar{a})d$$

$$v_o+\bar{k}-p(\bar{a})d \le v_o+\bar{k}-(a^*-\bar{a})-p(a^*)d. \qquad \text{(A6)}$$

$$V[v_o+\bar{k}-(a^*-\bar{a})-p(a^*)d] \ge V[v_o+\bar{k}-p(\bar{a})d] > \bar{V} \qquad \text{(A7)}$$

where the first unequality assumes V is monotonic and the second follows from (A5).

Substituting (A4) into (A7) yields

$$V^* = V[v_o+k^*] > \bar{V}. \qquad \text{Q.E.D.}$$

FOOTNOTES

1/ Large settlements in recent liability cases have caused premiums for liability insurance to increase sharply. In some cases coverage has been eliminated. In response, many firms are reducing or withdrawing the provision of certain goods and services because of the inability to secure liability coverage at a reasonable cost. See the Wall Street Journal, January 21, 1986, page 37.

2/ For example, liability under the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) is limited to clean-up costs, other costs of remedial actions, and damages to natural resources. In particular, it does not include damages to third parties. Dollar limitations on liability exist under other statutes as well, including the Price Anderson Act in the case of nuclear accidents and several laws governing marine pollution (See Section VIII).

3/ This internalization is not necessarily perfect. See Opaluch (1984) and Shavell (1984a, 1984b) for discussions of some problems associated with the use of liability rules to internalize externalities.

4/ We use the terms "clean-up costs" and "damages" interchangeably. However, in reality the two may be different and which one would be used as the basis for liability could vary from case to case.

5/ We assume that there is a single responsible party to avoid the problem of assigning liability and the potential for free-riding in the multiple polluter case. For a discussion of free-riding in principal-agent models, see Holmstrom (1982).

6/ Hereinafter, we use the term "full liability" rather than "strict liability" to refer to a liability rule where polluters must pay the full amount of damages. This is to avoid the potential confusion caused by the fact that it is possible to have "strict" (in the sense of "no-fault") liability for an amount that is less than total damages. Throughout the paper, our use of the term liability refers to no-fault liability, which may or may not be for the full amount of damages. In particular, we do not explicitly consider the negligence rule under which firms would be liable only if found to be negligent (although our treatment of regulation is similar to a negligence system under which firms never choose to be negligent). For a comparison of negligence and no-fault liability, see Shavell (1980, 1982).

7/ Because f is a contingency rule, i.e. it gives the level of liability contingent on a given realization of the random variable c, the first order conditions depend upon realized marginal utilities rather than expected marginal utilities. See Raiffa (1968) for a more detailed discussion.

8/ Although the expected utility hypothesis is the paradigm used most frequently in economic models of decision-making under uncertainty, it has been subject to a great deal of criticism by many economists, social psychologists and decision analysts. See, for example, the survey by Schoemaker (1982).

9/ The question of whether compliance with regulations issued pursuant to environmental statutes preempts common law used to impose liability is the subject of considerable debate. Some argue that regulatory standards simply provide a minimum set of standards for conduct, while others argue that the creation of a comprehensive regulatory program should be viewed as an attempt by Congress to provide a substitute for common law. The debate was fueled by the 1982 Supreme Court decision in the case of Milwaukee vs. The State of Illinois. (For a discussion of the concerns and issues regarding that decision, see U.S. Senate Hearing 98-247.) Recently, several bills have been introduced in Congress that would significantly reduce the liability of firms that are in compliance with regulations.

10/ This could be important if policy decisions are made by bargaining in the political arena and polluters have sufficient political clout to prevent the adoption of policies that would reduce their expected utility.

11/ See footnote 9.

12/ This is consistent with Shavell's (1982) result that, when victims are risk neutral and injurers are risk averse, a first best solution is possible under a negligence standard but not under strict liability. If a firm is only held liable when it is found to have violated the due care standard (assumed known by all), i.e. compliance with the standard implies lack of negligence, then with risk neutral victims the negligence system is equivalent to regulation requiring the due care level of precaution. Note, however, that Shavell compares strict

liability and negligence to a first best solution that only requires that a resource constraint be met in terms of expected value, not in each state. Thus, it is not really an ex post transfer problem such as that considered here, i.e. Shavell's first best solution is not the Pareto optimal solution in the absence of incentive problems considered in Section II which implicitly imposes a budget-balancing constraint in each state.

13/ See Schoemaker (1982) for a survey of issues.

14/ For a discussion of the dynamic nature of the legal system, see Blume and Rubinfeld (1982).

REFERENCES

1. Kenneth Arrow and Robert C. Lind, "Uncertainty and the Evaluation of Public Investment Decisions." American Economic Review 60:364-378 (1970).

2. Lawrence E. Blume and Daniel L. Rubinfield, "The Dynamics of the Legal Process." Journal of Legal Studies 11:405 (1982).

3. Jon M. Conrad, "Oil Spills: Policies for Prevention, Recovery, and Compensation." Public Policy 28:143-170 (1980).

4. Anthony C. Fisher, "Environmental Externalities and the Arrow-Lind Public Investment Theorem." American Economic Review 63(4):722-25 (1973).

5. Sanford J. Grossman and Oliver D. Hart, "An Analysis of the Principal-Agent Problem." Econometrica 51(1):7-45 (January 1983).

6. Bengt Holmstrom, "Moral Hazard and Observability." Bell Journal of Economics 10(1):74-91 (Spring, 1979).

7. _____, "Moral Hazard in Teams." Bell Journal of Economics 13(2):324-340 (Autumn, 1982).

8. Gary V. Johnson, Charles D. Kolstad and Thomas S. Ulen, "The Use of Both Ex Ante Regulation and Ex Post Liability Rules to Correct Environmental Externalities: Redundant or Optimal?" Institute for Environmental Studies Staff Paper No. 29, University of Illinois, December 185.

9. Richard E. Just and David Zilberman, "Asymmetry of Taxes and Subsidies in Regulating Stochastic Mishap." Quarterly Journal of Economics 43(1):139-148 (February, 1979).

10. Hayne E. Leland, "Optimal Risk Sharing and the Leasing of Natural Resources, with Application to Oil and Gas Leasing on the OCS." Quarterly Journal of Economics 42(3):413-437 (1978).

11. James J. Opaluch, "The Use of Liability Rules in Controlling Hazardous Waste Accidents: Theory and Practice.: Northeast Journal of Agricultural and Resource Economics 13(2):210-217 (October, 1984).

12. James J. Opaluch and Thomas A. Grigalunas, "Controlling Stochastic Pollution Events Through Liability Rules: Some Evidence from OCS Leasing." Rand Journal of Economics 15:142-151 (Spring, 1984).

13. Howard Raiffa, Decision Analysis: Introductory Lectures on Choice under Uncertainty. (Reading, Mass, 1963).

14. Stephen A. Ross, "The Economic Theory of Agency: The Principal's Problem." American Economic Review 63(2):134-139 (May, 1973).

15. Paul J.H. Schoemaker, "The Expected Utility Model: Its Variants, Purposes, Evidence and Limitations." Journal of Economic Literature 20(2):529-563 (June, 1982).

16. Steven Shavell, "Risk Sharing and Incentives in the Principal-Agent Relationship." Bell Journal of Economics 10(1):55-73 (Spring, 1979a).

17. _____, "On Moral Hazard and Insurance." Quarterly Journal of Economics 93(4):541-562 (November, 1979b).

18. _____, "Strict Liability vs. Negligence." Journal of Legal Studies 9(1):1-25 (January, 1980).

19. _____, "On Liability and Insurance." Bell Journal of Economics 13(1):120-132 (Spring, 1982).

20. _____, "A Model of the Optimal Use of Liability and Safety Regulation." Rand Journal of Economics 15(2):271-280 (Summer, 1984a).

21. _____, "Liability for Harm vs. Regulation of Safety." Journal of Legal Studies 13(2):357-374 (June, 1984b).

22. Joseph E. Stiglitz, "Risk Sharing and Incentives in Sharecropping." Review of Economic Studies, 61(2):219-256 (April, 1974).

23. U.S. Congress, Senate, Committee on the Environment and Public Works Extending and Amending the Clean Water Act, Senate Hearing 98-247.

24. Michelle J. White and Donald Wittman, "A Comparison of Taxes, Regulation, and Liability Rules Under Imperfect Information." Journal of Legal Studies 12:413-425 (June, 1983).