Marine Resource Economics, Vol. 5, pp. 1–21, 1988
Printed in the UK.

0738-1360/88 $3.00 + .00

Measuring Damages to Marine Natural Resources from Pollution Incidents under CERCLA: Applications of an Integrated Ocean Systems/Economic Model

THOMAS A. GRIGALUNAS
JAMES J. OPALUCH

Department of Resource Economics
University of Rhode Island
Kingston, RI

DEBORAH FRENCH
MARK REED

Applied Science Associates
Narragansett, RI

Abstract *Several pieces of federal environmental regulation establish strict liability for damages from spills of oil and hazardous substances. This paper discusses the Natural Resource Damage Assessment Model for Coastal and Marine Environments (NRDAM/CME), which is to be used for assessing damages from spills of oil or hazardous substances in coastal and marine environments under CERCLA and the Clean Water Act, as amended. The approach employs an integrated ocean systems/economic model to simulate the physical fates and biological effects of a spill and to measure the resulting economic damages. To illustrate application of the model, selected results are presented for hypothetical spills of a number of substances in a variety of environments. The results show that the damage function depends on the physical and chemical properties of the substance spilled, the season, and the environment in which the spill occurs.*

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 provide standard procedures for simplified assessments requiring minimal field observations, and type B regulations specify alternative protocols for conducting assessments in individual, site-specific cases (Sec. 301(c)(2)). Hence, the Act recognizes that damage asssessment 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 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 than 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 (National Research Council 1985: 424,432). For example, approximately $1.4 million (1986 dollars) 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; 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. Typically, approaches used by individual states to measure damage to natural resources rely on counts of dead fish, unit penalty calculations, or other approaches which can be ad hoc.[1] Although such approaches provide a simple procedure for assigning a dollar value following an incident, they do not provide a conceptually defensible methodology for measuring damages. Moreover, they do not meet the requirements of CERCLA which mandates that liability be based on damages—not on penalties.

This paper describes an alternative approach for measuring liability for natural resource damages from pollution incidents based on the concept of a damage function. The approach advanced is simple to use, in keeping with the requirements of CERCLA for those incidents which will result in use of a type-A natural resource damage assessment. However, the model developed—the Natural Resource Damage Assessment Model for Coastal and Marine Environments (NRDAM/CME or "the model")—is quite complex, and the data bases developed to implement the model are extensive. The model, which runs on the IBM PC (or compatible), 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. The framework described in this paper is the approach promulgated by the U.S. Department of Interior for use in measuring damages to coastal and marine environments for type-A natural resource damage assessment covered under CERCLA (43 CFR Part 11, 20 March 1987).

The remainder of this paper is organized as follows. Section II describes the NRDAM/CME and data in order to provide the reader with an appreciation of the critical interdisciplinary features of the analysis and the extensive data bases used. In section III, the model is applied to measure the damages from hypothetical oil and hazardous substance spills in selected coastal and marine environments. A summary and concluding comments are contained in section IV, and an appendix describes data sources and the development of information used in the model.

Overview of Model and Data[2]

Clearly, the consequences of a given oil or hazardous substance spill could vary greatly, depending upon the amount and physical and toxicological properties of the substance spilled, the characteristics of the environment in which the spill occurs (e.g., season, water depth, temperature), and the specific natural resources in the affected area and their value. To measure damages from a particular incident, linkage must 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 and associated data bases: the physical fates, biological effects, and economic damages submodels.

The physical fates submodel has a chemical data base which contains 14 parameters for 469 oil and chemical substances. The physical, chemical, and toxicological data contained in this data base include such parameters as density, solubility, vapor pressure, degradation rates in sea water and in sediments, 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 sediments. In addition, the submodel accounts for the amount of pollutant lost to the atmosphere through evaporation, where appropriate. A mass balance calculation ensures that the sum of the mass of the pollutant in all environmental compartments at each point in time equals the mass spilled.

To simulate the transport and 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, and the air temperature and distance to shorelines, or boundaries of concern, in each direction. In a particular application, these environmental parameters must be set by the user.

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 effects submodel, which calculates injury to various biota in the

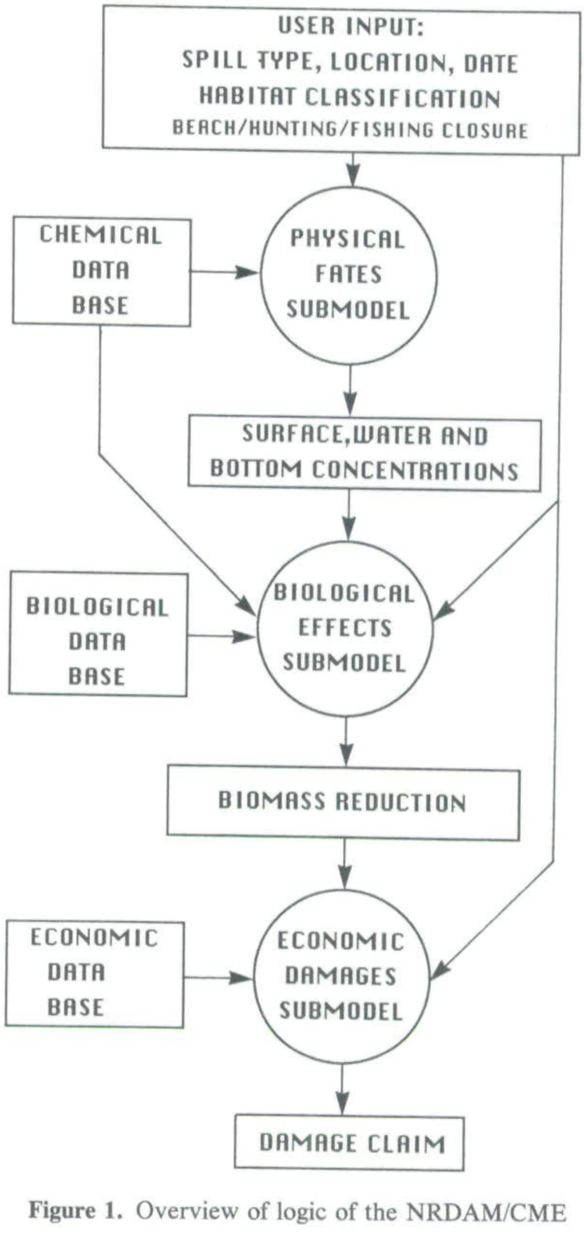

Figure 1. Overview of logic of the NRDAM/CME

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 (shorebirds, seabirds, and waterfowl). 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 varies by season, bottom type, marine vs. estuarine, and tidal vs. subtidal environments. In total, 364 different ecosystem-season categories are considered in the biological effects submodel.[3]

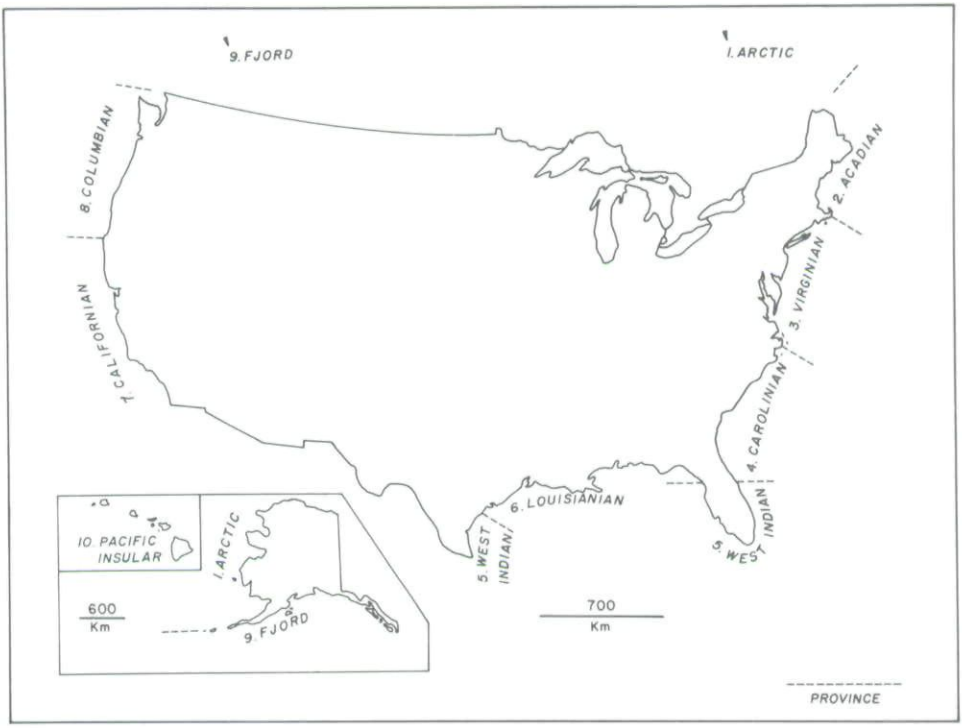

Figure 2. Boundaries of 10 Marine and Estuarine Provinces

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 quality-controlled toxicity test data. The biological submodel calculates direct loss of adults and juveniles for birds and for nine fish and shellfish species' categories and loss of larvae for each of the fishery categories. In addition, a simple ecological model is used to trace indirect losses through the food web.

Biological injury quantified in the submodel includes (1) short-term injury (e.g., death of adult fish) and (2) long-term injuries which occur over time (e.g., reduced recruitment from loss of larvae or juveniles). Only acute mortality is considered in the biological effects submodel; more complex changes, such as sublethal effects and alterations in food webs, are not considered because they require more data than currently are available.

Three categories of short-term biological effects are considered. First, surface slicks (e.g., oil) may be 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 also are taken into account. The dynamics of the biological system are traced using the Ricker model (1975). This model is used to simulate injuries to commercial and recreational fish and to birds. Because this model plays a central role in the analysis, it is described below in some detail, focussing on its application to the quantification of injuries and the measurement of damages to fisheries.

The Ricker model 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 of individuals within a particular age class are described as:

$$N(t) = N(0) \exp[-(M + F)\, t] \tag{1}$$

where N(t) is the number of individuals within an age class at time t, M is the natural mortality rate, and F is the fishing mortality rate.

The growth for an individual is determined by the Von Bertalanfy equation:

$$L(t) = L(\infty)\,(1 - \exp[-K\,(t - t_0)]) \tag{2}$$

where L(t) is length at age t, $L(\infty)$ is the maximum length, K is the Brody growth coefficient and t_0 is a constant (Ricker 1975). Weight is calculated from length using the equation:

$$W(t) = a\, L(t)^b \tag{3}$$

where W(t) is the weight at time t.

Thus, the dynamics of the biomass of a particular age class at time t is:

$$B(t) = N(t)^*W(t) = N(0)^*W(t)^* \exp[-(M - F)\, t] \tag{4}$$

where B(t) is the biomass of the cohort at age t. The total biomass of all cohorts in the fishery is:

$$X(t) = \sum_{i=I_r}^{I_{max}} B(i) \tag{5}$$

where X(t) represents the total biomass of all age classes in the fishery, I_r is the age class at the time of recruitment to the fishery (i.e., the youngest age class in the fishery), I_{max} is the age class at maximum life span for the species (oldest age class in the fishery).

Following the bioeconomics literature (e.g., Clark 1985) commercial catch, or harvest, at some point in time is assumed to be determined by:

$$H(t) = q\, E(t)\, X(t) \tag{6}$$

where H is harvest, q is the catchability coefficient, E is commercial fishing effort, and X is the total biomass. Thus, the instantaneous rate of commercial harvest at time t, defined as fishing mortality, is:

$$F(t) = q\, E(t).$$

The change in the *in situ* commercial value of the fishery at time t due to the spill is:

$$V^{NS}(t) - V^{S}(t) = [p^{NS}q^{NS}E^{NS}(t)X^{NS}(t) - C^{NS}E^{NS}(t)] - [p^{S}q^{S}E^{S}X^{S}(t) - C^{S}E^{S}(t)] \quad (7)$$

where the superscript NS represents the case with no spill and S represents the case with the spill, C is the cost per unit of effort, p is the price, and q and E are defined as in (6).

In general, it must be determined how all of the variables in (7) 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. Small spills also 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, although research is in progress on this issue (Opaluch 1988).

Under these assumptions, the lost *in situ* value of the commerical fishery at some point in time is:

$$V^{NS}(t) - V^{S}(t) = pqE(X^{NS}(t) - X^{S}(t)) = pF(X^{NS}(t) - X^{S}(t)) \quad (8)$$

where the superscripts are dropped for the variables which are assumed unchanged by the spill. The total discounted lost *in situ* value can be expressed as:

$$\int_{ts}^{T} \exp(-rt)\,[V^{NS}(t) - V^{S}(t)]\,dt \quad (9)$$

where t_S is the time of the spill, T is the time at which the fishery is completely recovered[4] and r is the social discount rate. Substituting from (8), the total discounted loss can be expressed as:

$$\int_{ts}^{T} \exp(-rt)\,pF(X^{NS}(t) - X^{S}(t))\,dt. \quad (10)$$

The treatment of recreational fisheries is formally equivalent to that for commerical fisheries. The total discounted value of the recreational fishery from a spill is:

$$\int_{ts}^{T} \exp(-rt)\,P_{rec}F_{rec}\,(X^{NS}(t) - X^{S}(t))\,dt \quad (11)$$

where P_{rec} is a unit loss in consumer surplus per unit loss in recreational catch and F_{rec} is the recreational fishing mortality rate.

Using the methodology described above, long-term commerical and recreational fishery losses due to the effects of acute toxicity on the biomass are considered. The output of the biological effects submodel is a time series of lost catch for each of the nine species groups for fin and shellfish, as well as losses in various groups of birds and fur seals.[5]

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 web organisms that have no commercial or recreational value in themselves but that contribute to consumer or predator species which do have economic value.

Conceptually, destruction of lower trophic biota constitutes damages to organisms which, in effect, are inputs to the production of marine life that may have commercial or recreational value (see, e.g., Ragozin and Brown 1985; Kahn and Kemp 1985). Hence, the demand for the services of lower trophic species is derived from the value of the species ultimately produced by lower trophic biota. In the model, damages resulting from injury to lower trophic, non-commercial organisms are based on the ultimate loss in the *in situ* use value of consumer species (commercial and recreational fisheries, waterfowl, shorebirds, seabirds, and fur seals) which occurs when an incident affects the productivity of the food web.[6]

A food web or ecological model developed in the biological effects submodel 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, shorebirds and seabirds and to fur seals, outlined in this paper.

Once the short-term and long-term biological injuries have been quantified following a particular incident, damages can be measured. As noted, 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.

In order to measure lost *in situ* use value, fish resources injured by an incident must be allocated between commercial and recreational harvests foregone. 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 given an estimate of the total fishing mortality rate for species groups. Total fishing mortality is broken into commercial and recreational fishing mortality as:

$$F_{TOT} = F_{COM} + F_{REC} = q_{COM}E_{COM} + q_{REC}E_{REC} \tag{12}$$

and commercial and recreational fishing mortality rates can be calculated as:

$$F_{COM} = [H_{COM}/(H_{COM} + H_{REC})]F_{TOT} \tag{13}$$

and

$$F_{REC} = [H_{REC}/(H_{REC} + H_{COM})]F_{TOT} \tag{14}$$

Hence, given estimates of total fishing mortality rates and commercial and rec-

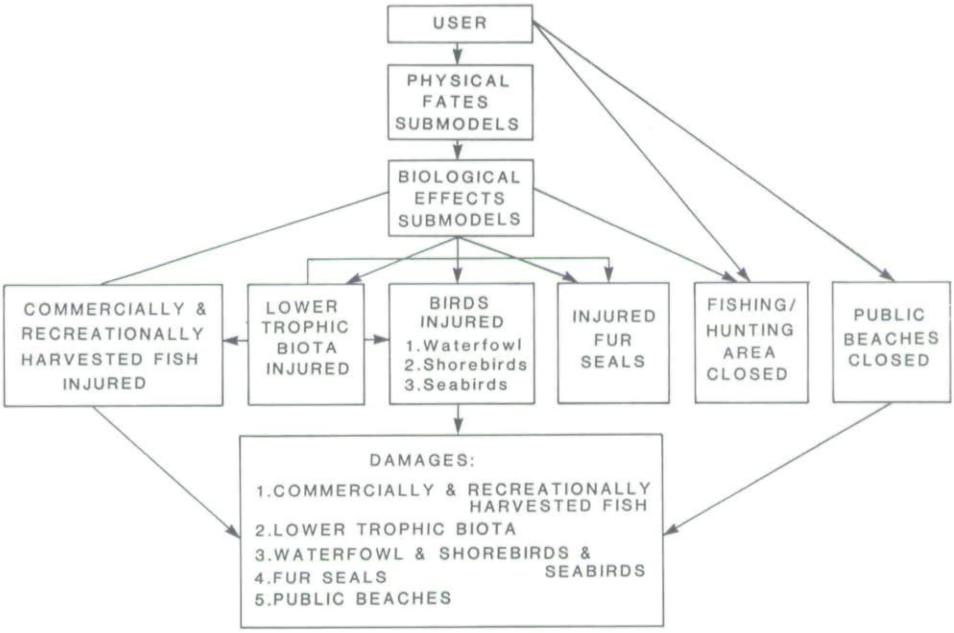

Figure 3. Simplified representation of the natural resource damages assessment process and the damage categories considered in the economic damages submodel of the NRDAM/CME

reational catch, lost stock can be allocated among lost recreational and commercial catch over time as:

$$H^{NS}_{COM} - H^{S}_{COM} = q[H^{NS}_{COM}/(H^{NS}_{COM} + H^{NS}_{REC})]F_{TOT}(X^{NS} - X^{S})$$

$$H^{NS}_{REC} - H^{S}_{REC} = q[H^{NS}_{REC}/(H^{NS}_{COM} + H^{NS}_{REC})]F_{TOT}(X^{NS} - X^{S})$$

Ex-vessel (price at the dock) real fish prices by species, averaged over 1982-1985, 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 (NMFS) sources. To provide a measure of damages from lost biomass of sports-caught fish, estimates of marginal value per fish (Norton et al. 1983; Rowe 1985) were divided by the average weight of sports-caught fish of the particular species in the region considered. The average weights by species were obtained from NMFS sources (1984,1985b).

Injury to waterfowl, shorebirds and seabirds 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 submodel, using the same Ricker-type cohort model as for fisheries, described above. Damages resulting from consumptive use value losses are measured using available estimates of the marginal value of an additional waterfowl (duck or geese) harvest (Hay and Charbonneau 1974). 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 is evaluated based on a unit day value published by the Water Resources Council (1979).

The real discount rate used is 10%, the rate specified for use by the Office of Management and Budget Circular A-94, as revised.

Discussion of Selected Results

This section describes applications of the NRDAM/CME to a variety of environments and substances spilled. The estuarine spills presented in this section assume that all incidents take place at a location where the water depth is 30 feet, and the pycnocline (separating the upper and lower water columns) is assumed to be at 15 feet. Except where otherwise indicated, all spills are assumed to take place on mud bottoms during the summer season when the air temperature is taken to be 24 degrees C. It also is assumed that none of the spilled material is cleaned up.[7]

The sample runs were chosen to illustrate how the results change as the major characteristics of hypothetical spills vary. Crude oil and diesel fuel oil are by far the most common substances spilled, and the base case incident is a 100 metric ton (735 bbl) diesel oil spill in the Virginian Province. Sensitivity analyses were run in which the quantity of diesel oil spilled varied from 5 metric tons (36.8 bbl) to 1,000 metric tons (7,350 bbl) in both marine and estuarine environments to indicate how damages vary with the quantity spilled in these two different environment-types. Additional cases were run to examine how damages vary as a function of the province in which the spill occurs and the season of the spill. For this purpose, the base case 100-metric-ton, estuarine, diesel-oil spill was run in all ten provinces during each of the four seasons. Also, a 100-metric-ton diesel-oil spill was run for several intertidal cases to illustrate the extent of damages from spills which affect the shoreline.

Additionally, a series of runs is provided for different oils and refined products as well as non-oil products to examine 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. Each of the non-oil cases is run for spills of 50 and 100 metric tons.

Base Case Results

Table 1 contains the model output for the base case, 100-metric-ton, diesel-oil spill. As can be seen, the largest categories of damages to finfish occur to demersal fish (e.g., flounder) at $13.3 thousand, piscivorous fish (e.g., striped bass, bluefish) at $12.6 thousand, and semi-demersal fish (e.g., cod) at $11.3 thousand. The damages to mollusks (e.g., oyster, clams, scallops) total $4.2 thousand, and damages to decapods (e.g., crabs) are minimal at a total of $63. Also, some damages occur to seabirds and waterfowl ($1.1 thousand). Had the spill occurred during the fall or winter season, when waterfowl is much more abundant, this category of damages would have been considerably higher. In summary, the present value of the total damages from this 100-metric-ton spill is $47 thousand.

Table 1
Economic Damages from a 100-Metric-Ton Spill of Diesel Fuel Oil in a Mud Bottom, Estuarine Environment of the Virginian Province During the Summer Season (Expressed in 1986 dollars).

Category	*In Situ* Losses
Commercial and Recreational Finfish	
Anadromous fish	$ 71.82
Planktivorous fish	4,431.65
Piscivorous fish	12,584.98
Top carnivores	0.00
Demersal fish	13,303.25
Semi-demersal fish	11,345.97
Subtotal	$41,737.
Commercial invertebrates	
Mollusks	$ 4,153.40
Decapods	62.77
Squid	38.54
Subtotal	$ 4,254.
Birds and Mammals	
Fur seals	0.00
Sea birds	$ 295.25
Waterfowl	763.43
Subtotal	$ 1,058.
Summary:	
Fishery losses	$45,993.
Birds and fur seals	1,058.
Total damages	$47,051.

Damages as Function of Quantity Spilled

The results for the damages as a function of the quantity of diesel oil spilled in July in the estuarine and marine environments of the Virginian province are presented in Table 2 and in Figures 4 and 5. As indicated, beyond a threshold-size oil spill, damages increase with quantity spilled and are approximately linear as quantity spilled increases. A spill of 367 barrels of oil in an estuary leads to losses of $24 thousand. Increasing the quantity spilled by a factor of five to 1,837 barrels leads to damages of nearly $114 thousand. Hence, for this five-fold increase in amount spilled in an estuary, damages increase by a factor of 4.75.

In the marine environment, the damage function exhibits a similar shape, but damages are far smaller in absolute value, as expected. This occurs because of the lower levels of productivity, the greater water depths of the marine environment, and the absence of some species (e.g., shorebirds and waterfowl) in offshore areas.

Thus, damages from oil spills increase with the quantity spilled, such that the damage function is roughly linear. This suggests that, beyond a minimal spill size,

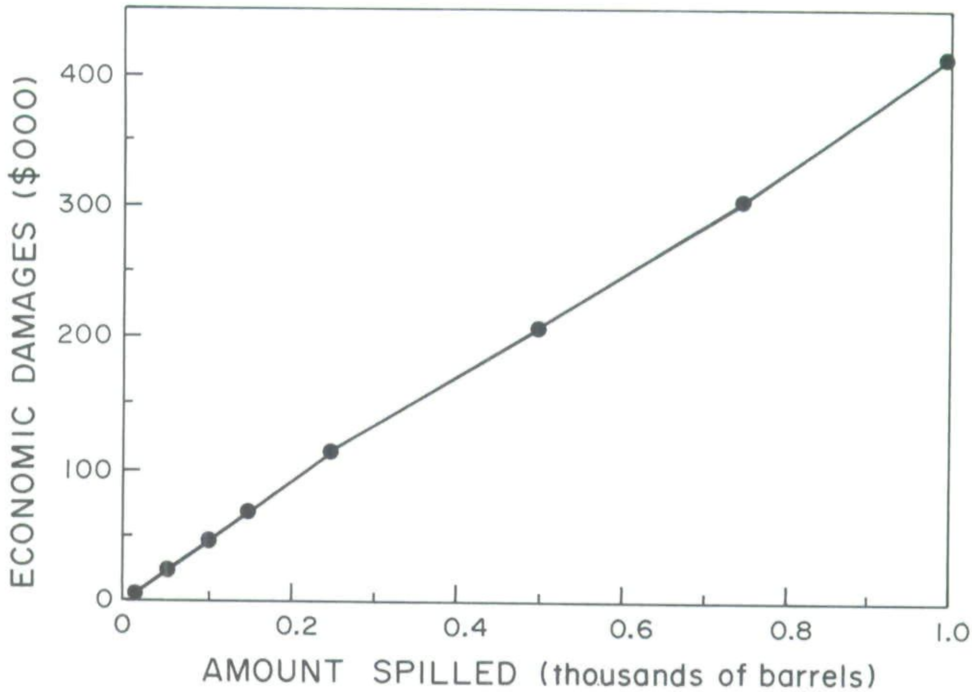

Figure 4. Damages from spills of diesel fuel in the estuarine environment of the Virginian Province

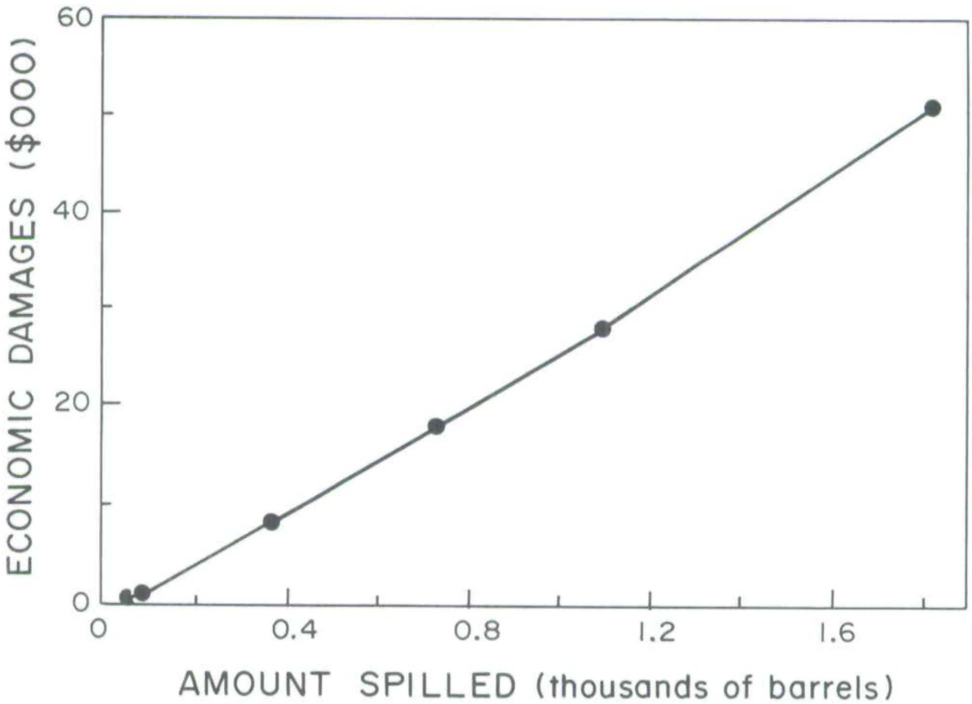

Figure 5. Damages from spills of diesel fuel in the marine environment of the Virginian Province

Table 2
Economic Damages from Spills of Diesel Fuel in the Virginian Province for Various Quantities Spilled During the Summer Season (Expressed in 1986 Dollars)

Quantity Spilled		Damages in Estuarine Environments	Damages in Marine Environments
Metric Tons	Barrels		
5	36.75	$ 2,491.	$ 329.
10	73.50	$ 4,742.	$ 944.
50	367.50	$ 24,373.	$ 8,263.
100	735.00	$ 47,050.	$ 14,300.
150	1,102.50	$ 69,297.	$ 28,033.
250	1,837.50	$114,007.	$ 51,285.
500	3,675.00	$210,400.	$134,635.
750	5,512.50	$310,972.	$211,375
1,000	7,350.00	$426,668.	$312,377.

a fixed charge per barrel spilled may be appropriate for oil spills. However, it is stressed that the threshold for damages and the unit charge vary from location to location and, indeed, varies by season for a given location.

Base Case Diesel Fuel Spill in Different Provinces

Damages from the base 100-metric-ton diesel oil spill in an estuary environment in each of the ten provinces are presented in Table 3. As can be seen, the highest level of damages occurs in the California province during the spring season ($373 thousand); the lowest damages are realized by a winter spill in the Arctic ($1,216).

In general, the results reveal major differences in the measure of damages

Table 3
Economic Damages from 100 Metric Ton Spills of Diesel Fuel in Estuarine Environments by Province and Season. (Expressed in 1986 Dollars)

	Spring	Summer	Fall	Winter
Acadia	$ 19,649.	28,429.	19,211.	14,723.
Virginia	$ 25,907.	47,049.	29,495.	21,599.
Carolina	$155,532.	214,832.	73,256.	101,491.
Louisiana	$181,142.	260,555.	105,506.	217,420.
West India	$ 21,570.	5,914.	10,724.	21,042.
California	$373,341.	168,699.	52,241.	367,887.
Columbia	$ 57,655.	190,777.	80,540.	94,230.
Fjord	$ 41,208.	49,411.	39,295.	19,500.
Arctic	$ 65,095.	106,535.	48,205.	1,216.
Pacific Insular	$ 30,403.	6,852.	1,705.	4,450.

Table 4
Economic Damages from 100-Metric-Ton Spills of Diesel Fuel Washing Ashore on Various Shoreline Types in the Virginia Province During the Summer Season (Expressed in 1986 Dollars)

Shoreline Type		Economic Damages
Rock		$447,018.
Cobble		$114,759.
Mud Flat		$ 11,466.
Salt Marsh		$ 79,723.
Sand		
Biological Damages	$8,042.	
Beach Damages	$7,253.	
Total Damages for Sand		$ 15,295.

across provinces and seasons from an identical spill. This is as one would expect, given the significant differences that exist from province to province in terms of their biological resources and their economic value and given the seasonal variations in biological resources within a province. The high damage estimates for southern provinces during the winter and spring reflect the large numbers of birds which winter in southern areas. This seems consistant with experience with spills in these areas during these seasons. For example, the *San Joaquin Valley* crude oil spill which occurred off the coast of California in January 1986 resulted in bird losses per barrel spilled of two to three orders of magnitude higher than other spills, such as the *Argo Merchant*, the *Amoco Cadiz*, *Puerto Rican*, and the *Arco Anchorage* spills. The results for marine areas (not presented here) reveal a similar wide variation across provinces and from season to season, although for most provinces, damages in the marine environment in a given season are less than for the estuarine environment.

Intertidal (shoreline) Spills

Table 4 contains the results of spills of 100 metric tons of diesel oil striking the intertidal area in the Virginian province. Five intertidal environment types are considered. An intertidal spill on a sandy shore results in damages of $15.3 thousand, of which $8 thousand is attributable to biological (fish, shellfish, and birds) damages and $7.3 thousand is a result of damages from a presumed closure of 100 meters of non-federal public beaches.[8] For the other intertidal environment types, damages range from $447 thousand for a rocky shoreline to $11 thousand for a mud flat. Again, the results demonstrate that the magnitude and type of damages from a given spill are very sensitive to the type of environment affected. The high level of damages in the rocky shoreline in the Virginian province, for example, reflects very high concentrations of mussels in this environment.

Other Oil, Refined-Petroleum Product, and Non-Oil Spills

Results for a variety of oil, refined-petroleum products, and non-oil substances serve to illustrate how damages vary as a function of the physical and chemical

Table 5
Economic Damages from 100-Metric-Ton Spills of Various Petroleum Products in an Estuarine Environment in the Virginia Province During the Summer Season (Expressed in 1986 Dollars)

Substance	Economic Damages
Crude Oils:	
Heavy Crudes:	
Prudhoe Bay Crude Oil	$132,069.
Cook Inlet Crude Oil	$124,262.
Medium	$149,854.
Light	$ 90,213.
Refined Products:	
Gasoline	$113,210.
Kerosene	$ 31,484.
No. 2 Fuel Oil	$179,378.
No. 6 Fuel Oil	$ 46,267.

properties of the material spilled. Again, all spills in this analysis are assumed to occur in a Virginian estuarine environment during the summer season.

In the top part of Table 5, damage results are presented for selected crude oils and refined-petroleum products, assuming a 100-metric-ton spill. Each of these substances differs with respect to its physical and chemical properties (e.g., density, solubility, and evaporation rates), and the damage measure reflects these differences. For example, among the crude oils, the heavier oils cause considerably higher damages than a light crude oil spill, in large part because bird losses from slicks will be greater with a heavier crude oil as compared with a light crude oil. Considerable differences also exist among the damages resulting from refined products spills, because of the differences in their physical and chemical properties.

Lastly, results for damages from selected non-oil substances are illustrated (Table 6). For example, in contrast with oils and refined-petroleum products, which float or are soluble, a sinking substance is pentachlorophenol (PCP). PCP

Table 6
Economic Damages from Spills of Non-Petroleum Substances in Virginia Estuaries in the Summer Season (Expressed in 1986 Dollars)

Substance	Damages @ 20 Metric Tons	Damages @ 100 Metric Tons
Pentachophenol	$197,258.	$471,850.
Aldrin	$427,803.	$968,474.
Benzene	$ 25,795.	$104,014.
Xylene	$ 4,096.	$ 23,950.
Toluene	$ 11,506.	$ 51,674.

has a density of 1.98, is relatively insoluble, has a half-life of about 90 days in seawater, and is quite toxic (the LC50 is 115 ppb for fish). For a 20-metric-ton spill of PCP, total damages are $197 thousand; and, for a 100-metric-ton spill, damages total approximately $472 thousand. Hence, damages increase with the amount of PCP discharged; but, in contrast with the case for diesel fuel oil presented in Table 2, damages increase much less than proportionately with the amount spilled. 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 and refined products tend to spread out relatively quickly, covering a substantially larger area as the amount spilled increases. The range of damages revealed in Table 6 for the different non-oil substances illustrates the sensitivity of damages to the various characteristics of the specific material spilled.

IV. Summary and Concluding Comments

As indicated at the outset of this paper, the liability provisions of CERCLA, the CWA, and other federal legislation are potentially very significant for environmental policy. Although specific provisions of the Acts differ (see, for example, Opaluch 1984), they provide a consistent national framework not only for compensating for damages but also for invoking what is akin to a Pigovian tax on polluters. However, as has been widely recognized, the measurement of damages from pollution incidents clearly involves extremely complex issues encompassing several disciplines in addition to resource and environmental economics. Hence, attempts to develop operational frameworks for implementing policies to internalize the social cost of pollution present formidable challenges, particularly in the marine environment where many inherent problems make it very difficult to observe biological injuries following a spill.

This paper has described the type A approach for simplified natural resource damage assessments under CERCLA. The NRDAM/CME and its data bases were briefly described, and the application of the model was illustrated using a variety of hypothetical spills as sensitivity analyses. By simulating the dispersion and degradation of a spilled substance in the environment, quantifying the resulting short- and long-term biological injuries, and providing a measure of economic damages, the NRDAM/CME obviates the need for expensive field observations, which would be inappropriate for type-A simplified assessments.

The example results presented show that the 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 for spills in coastal and marine environments must consider the characteristics of the substance and 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 a sinking substance release, PCP, where the area impacted is more restricted. For the latter case, releases in the range of 20-100 metric tons resulted in decreasing average damages per ton of PCP as the amount released increases. In contrast, average damages per barrel of oil discharged increase almost linearly with the amount spilled.

The results also show that, for a given spill, damages vary considerably depending upon the environment-type (e.g., estuary vs. marine and subtidal vs. intertidal), the province, and the season. It follows that an approach which assigns liability to spills without regard to considerations such as the environment, location, and season can easily be in error concerning the relationship between damages and quantity released or other characteristics of the pollution incident.

By way of conclusion, the use of the NRDAM/CME to determine liability for spills under CERCLA is a new approach and is novel in its use of newly available computer technologies as part of a regulatory framework. As such, use of the NRDAM/CME raises a number of issues. A central question is the extent to which one can validate the NRDAM/CME, for example, by comparing the results of case studies of spills with those obtained through use of the model. A second set of issues involves policy concerns which arise in actually implementing the type A approach described in this paper and, more generally, damage assessments under CERCLA, of which the type A approach is only a part. For example, how do the concepts in the NRDAM/CME, which are based on available data and necessarily simplified assumptions, conform to those specified in CERCLA—and to those that welfare economists typically consider in dealing with social costs? How can the integrity of an approach designed for use by state and federal trustees—but readily available for use by all—be maintained? Will the type-A damage assessment approach, in fact, provide incentives to polluters to avoid spills? A final issue mentioned here concerns uses of the NRDAM/CME for purposes other than determining liability *ex-post*. For example, how might one use the model as an *ex-ante* risk assessment planning tool to evaluate the potential environmental costs of a proposed natural resource policy, such as offshore oil development or waste disposal at sea? These (and other) issues are addressed in forthcoming papers by the authors.

Acknowledgements

This work was carried out while Dr. Grigalunas and Dr. Opaluch were on leave of absence with Economic Analysis Inc., Wakefield, RI. The work reported on in this paper was funded by the U.S. Department of the Interior. Dr. Grigalunas and Dr. Opaluch also acknowledge support by the Rhode Island Agricultural Experimental Station for research on issues relating to environmental policy (AES Contribution 2404). The views are those of the authors and do not necessarily reflect the views of the funding agencies. The authors benefited from comments received from an anonymous referee.

Appendix I. Discussion of Data

The development of the NRDAM/CME required that substantial data bases containing biological, economic, and physical/chemical/toxicological information be assembled. This appendix summarizes the development of these data bases. Readers interested in a more detailed discussion of the data and its sources are referred to the original study document (Economic Analysis, Inc. and Applied Science Associates, Inc., 1987).

Biological Data. To make the data gathering process a tractable task, an areal approach was used, as outlined in Cowardin et al. (1979). The hierarchical system used in this publication divides a given province into marine and estuarine systems; and, within each of these systems, subtidal (always submerged) and intertidal (at times submerged) subsystems are considered. Finally, within each subsystem, a variety of classes of bottom types are enumerated (e.g., rock bottom or shore, reefs, unconsolidated bottom, and aquatic bed).

Of the 340 possible combinations of province-system-subsystem-class, only 224 were judged to actually exist (e.g., there are no mangroves in Alaska). Groupings were made within the hierarchical system where certain combinations were determined to be approximately equivalent. This approach ultimately yielded 36 intertidal and 55 subtidal ecosystem types, for a total of 91 environments to be considered.

For each of the 91 ecosystem types, four categories of data were gathered: 1) adult biomass, by species; 2) larval numbers, by functional group; 3) mortality and growth rates by functional groups; and 4) productivity values. Data concerning adult biomass for all commercially and recreationally important species of fish and invertebrates were gathered and converted to a standing stock figure expressed in grams wet weight/square meter. Birds and fur seals were expressed as number/square km. Larval numbers, where available, were gathered by species as number per square meter. Mortality and growth rates were collected or derived by species and province, and a representative (mean) value determined for each category. Productivity values were collected for three trophic groups: primary, zooplankton, and benthos. Biomass and productivity data are reported for each season, in order to account for migrations and for larvae and productivity variability.

The primary sources of data for fish stocks and for mortality and growth rates and related information was the National Marine Fisheries Service (NMFS) of the National Oceanic and Atmospheric Administration. Estuarine and Intertidal data came from a variety of sources, including government and academe (e.g., Nixon 1983). In most cases, data were taken from the two to four largest or most important (commercially) estuaries in a province and averaged. These values, then, would reflect values for those areas where spills would be most likely to occur.

Waterfowl, shorebird, and seabird abundance data were obtained from the U.S. Fish and Wildlife Service, the available literature, and personal communications with experts. Population data for fur seals were extracted from the relevant literature.

Economics Data. The *in situ* value per pound of a particular species category will vary by season and environment, depending upon the species composition of that category. Hence, separate commercial and recreational price indexes were constructed for each of the nine species groups for as many as eight different bottom types, for estuaries and marine environments, intertidal and subtidal zones in all four seasons for each of the ten regions. This implies that a maximum of about 23,000 price indexes is required. However, since some bottom types do not exist in all provinces and since some environments have identical species compositions as other environments within the same province, some environmental types can be eliminated. By eliminating non-existing environments and replications, the total number of price indexes can be reduced to 8,856. For any given spill, the model needs to choose the 18 indexes which reflect the species composition of the commercial and recreational losses for the nine species groups in the environment and season of concern.

The principal commercial values for categories of finfish and shellfish are the NMFS data on total weight and value of catch by species and region for the years 1982-85. In the case where a species is contained in the biological data submodel but catch is not reported in the NMFS data, NMFS statistics for "other finfish" or "other shellfish" are used, as appropriate. For recreational fisheries, the principal data used are the total number and weight of sportsfish caught by species and subregion (NMFS 1984, 1985) and the value of changes in catch rates for sportsfishing, based on estimates in the literature (e.g., Norton et al. 1983 and Rowe 1985). Since comparable recreational data do not exist for recreational shellfishing, all fishing mortality for mollusks, decapods, and squid is presumed to represent commercial catch. Thus, the share of damages from injury to these categories which represents lost *in situ* recreational value is measured using commercial values. Finally, for environments where the biological data base has zero abundance for all species within a category, the economic data base uses the average of the price indexes for that category from all environments within that province.

An analysis of National Seashore facility and visitor data, State Comprehensive Outdoor Recreation reports, and an extensive telephone survey of appropriate local and state public officials was used to acquire an extensive data base concerning the availability and use of federal, state and other public beaches. Information concerning the non-market value of beach use was adapted from the relevant published and unpublished literature.

Chemical and Toxicological Data Base. This data base contains 14 parameters for almost 500 substances. For each chemical in the data base, the physical/chemical properties supplied include density, solubility, molecular weight, vapor pressure, adsorbed/dissolved partitioning coefficient (koc), and degradation rates (in water and in sediments). The toxicological data supplied include EC50 values for phytoplankton, LC50 values for phytoplankton, zooplankton, fish, benthos, and icthyoplankton (larvae fish and eggs) in parts per billion (ppb), and a threshold effects value (in ppb). The pertinent data for many parameters were extracted from various already-established data bases, such as the Oil and Hazardous Materials Technical Assistance Data System prepared by the Environmental Protection Agency. In some cases, parameter values had to be estimated using approaches established in the literature and available descriptive information.

Notes

1. A review of approaches used by several states to assign liability for marine-related natural resource damages from pollution incidents may be found in U.S. Dept. of Commerce, National Oceanographic and Atmospheric Administration (1984).
2. Due to space constraints, the discussion in this paper is only a brief version of an extensive technical report to which the interested reader is referred (Grigalunas, Opaluch, Reed, and French 1987).
3. An appendix to this paper describes data sources and development of the extensive information base used in the model.

4. Note that the system is always assumed to recover to the initial equilibrium. For two reasons, the model does not consider the potential for the hysteresis effect, where the system returns to an equilibrium which is qualitatively different from the initial state of the system. First, since the model is meant to be applied to "minor" incidents, it is unlikely that hysteresis would be a factor for the vast majority of incidents to which the model would be applied. Secondly, it would be impossible to predict cases where hysteresis would occur, or the nature of the subsequent equilibrium, particularly within the broad national framework for the Type A damage assessment process. For cases where hysteresis is an important concern in damage assessment, an incident-specific Type B analysis may be required.
5. Because LC50 data comparable to that for fish do not exist for birds and for fur seals, a different approach had to be used to quantify acute mortality from a slick to these species. Based on a review of the literature, the biological effects model assumes that 53% of birds and 63% of fur seals contacted by a slick die.
6. Lost primary (phytoplankton, benthic algae, and higher plants) and secondary (zooplankton) productivity is translated to lost yield for each animal species via reduced food supply using a simple food web model. The model is incident-specific and assumes that the share of the prey a predator consumes is proportional to its biomass relative to all other predators of that prey in the affected environment. Total lost production of a predator species is then equal to the sum of its lost food consumption times an ecological efficiency factor, which is the ratio of net growth in weight to weight of food assimilated. For a discussion of primary production and food chain dynamics, see Ryther 1969.
7. The physical fates submodel allows the user to specify the amount of the spilled substance cleaned up and the date of cleanup. Damages to natural resources are measured taking into account the amount of the substance recovered via cleanup efforts and the timing of removal operations. However, the results in this paper assume that no cleanup of spilled substances occurs in order to examine the maximum biological injuries and damages which could result from an incident.
8. Because of space constraints, the approach used to measure public beach damages in the NRDAM/CME is not presented in this paper but can be found in Grigalunas, Opaluch, Reed, and French (1987). Briefly, the economics data base contains information on public beach use per meter of beach, by month for federal and non-federal beaches in each province. Hence, when a spill leads to closure of a given length of public beach for a specific period of time, the number of lost trips can be estimated. Each lost trip is valued using an average of the estimate of consumer surplus per saltwater beach visit available in the literature.

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* 10, No. 2 (June): 125–137.

Brown, G. M., and J. Hammack. 1977. Commonweath v. Steuart, Economic Valuation of Waterfowl. Mimeo.

Clark, C. W. 1985. *Mathematical Bioeconomics: The optimal management of renewable resources*. New York: John Wiley & Sons.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe. 1979. *Classification of wetlands and deepwater habitats of the United States*. U.S. Dept. of Interior: Fish and Wildlife Service.

Grigalunas, T. A., R. C. Anderson, G. M. Brown, Jr., R. Congar, N. F. Meade, and P. E. Sorensen. 1986. Estimating the cost of oil spills: Lessons from the AMOCO CADIZ incident. *Marine Resource Economics* 2 (3):239–262.

Grigalunas, T. A., J. J. Opaluch, D. French, and M. Reed. 1987. *Measuring Damages to Coastal and Marine Natural Resources: Concepts and Data Relevant for CERCLA Type A Damage Assessments*. Springfield, VA: National Technical Information Service.

Hay, M. J., and J. Charbonneau. 1974. *Estimating the marginal value of waterfowl for hunting*. Draft. Washington, D.C.: U.S. Department of the Interior, Fish and Wildlife Service.

Kahn, J. R., and W. M. Kemp. 1985. Economic losses associated with the degradation of an ecosystem: The case of submerged aquatic vegetation in Chesapeake Bay. *Journal of Environmental Economics and Management* 12(3): 246–263.

Norton, V., T. Smith, and I. Strand. 1983. *Stripers: The economic value of the Atlantic coast commercial and recreational striped bass fisheries*. Univ. of Maryland Sea Grant Publication No. UM-SG-TS-83--12.

Opaluch, J. J. 1984. The use of liability rules in controlling hazardous waste accidents: Theory and practice. *Journal of the Northeast Ag and Resource Economics Assoc.* 13, No 2 (October).

Opaluch, J. J., and T. A. Grigalunas. 1984. Controlling stochastic pollution events through liability rules: Some evidence from OCS leasing. *Rand Journal of Economics* 15, No. 1 (Spring): 142–151.

Opaluch, J. J. 1988. Modeling behavioral response within a CERCLA Type A model. Dept. of Resource Economics, Univ. Of Rhode Island, Working Paper.

Ragozin, D. L., and G. M. Brown. 1985. Harvest policies and nonmarket valuation in a predator-prey system. *Journal of Environmental Economics and Management* 12(2):155–168.

Ricker, W. E. 1975. Computation and interpretation of biological statistics of populations. *Bull. Fish. Res. Board of Canada* 191(382).

Rowe, R. W. 1985. *Valuing marine recreational fishing on the Pacific coast*. National Marine Fisheries Service Administrative Report No. LJ-8-18C(June).

Ryther, J. H. 1969. Photosynthesis and fish production in the sea. *Science* (Oct. 3):72–76.

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.

U.S. Congress. 1980. *Comprehensive environmental response, compensation and liability act of 1980*. Public Law 96-510, 96th Cong. (11 Dec. 1980). Stat. 2767–2811.

U.S. Department of Commerce. National Marine Fisheries Service. 1985A. *Marine recreational fisheries statistics survey, Atlantic and Gulf coasts, 1983–1984*. Current Fisheries Statistics No. 8325. Washington, D.C., August.

U.S. Department of Commerce. National Marine Fisheries Service. 1985b. *Marine recreational fisheries statistics survey, Pacific coast, 1983–1984*. Current Fisheries Statistics No. 8326. Washington, D.C., August.

U.S. Department of Commerce. National Oceanographic and Atmospheric Administration. 1984. *The use of economic analysis in valuing natural resource damages*. Washington, D.C., June.

Water Resources Council. 1979. Unit day value method. *Federal Register* 44, no. 242 (Dec. 14).