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ANALYSIS

Policy evaluation of natural resource injuries using habitat equivalency analysis

Brian Roach [a,*], William W. Wade [b]

[a] Global Development and Environment Institute, Tufts University, 44 Teele Ave., Medford, MA 02155, United States
[b] Energy and Water Economics, Suite 304, 39 Public Square, Columbia, TN 38401, United States

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Abstract

The natural resources managed by government agencies are commonly subject to injuries from accidental events. In order for agencies to evaluate alternative management plans, economic damage estimates are required of potential natural resource injuries under alternative scenarios. However, accurate damage estimates are often difficult to obtain because of a lack of data on the ex ante economic costs of natural resource injuries. In recent years, trustees have increasingly used habitat equivalency analysis (HEA) to scale compensation for natural resource injuries. Unlike traditional economic analysis, which bases damage estimates on losses to human use (and sometimes nonuse) values, HEA estimates the ecological service loss of the injury and then scales restorative ecological compensation to offset these losses. Thus, HEA aims to maintain a baseline level of ecological functioning rather than a baseline level of human welfare.

This paper describes the first attempt to use the HEA approach as an ex ante policy evaluation tool. The specific policy application is offshore oil development managed by the U.S. Minerals Management Service. We describe the reasons HEA was deemed the appropriate methodology to evaluate the ecological damages of potential oil releases into the environment. We then discuss the procedures used to estimate potential natural resource injuries, derive suitable ecological compensation in a HEA framework, and convert restorative ecological compensation into economic damage estimates. The validity of the economic estimates is explored by comparison to existing data. We conclude that HEA offers a viable alternative to traditional economic analysis when potential injuries to ecological habitats are being evaluated.

Keywords: Habitat equivalency analysis (HEA); Cost–benefit analysis; Natural resource injuries; Natural resource damage assessment (NRDA)

1. Introduction

Natural resource agencies commonly manage resources that are subject to injury from accidental events. Alternative management plans evaluated by such agencies may entail different expected probabil-

* Corresponding author. Tel.: +1 617 627 6787; fax: +1 617 627 2409.
E-mail address: brian.roach@tufts.edu (B. Roach).

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ities of accidental natural resource injuries. Within a standard cost–benefit analysis (CBA) framework, alternative plans are assessed by multiplying the expected probability of a particular injury by the economic damages, measured in dollars, which would result from the injury. Policy analysis thus requires valid information on both the probabilities of accidental events and the associated economic damages.

The economic damages arising from natural resource injuries generally include reductions in both human use values, such as recreation or commercial activities, and nonuse, or passive use, values. Estimates of human use damages are obtained or revealed from market behavior, with dollar values derived from market prices, travel cost models, or similar methodologies. These approaches are widely accepted and are commonly incorporated into CBAs. However, monetary estimates of nonuse damages have typically relied on stated preference approaches, specifically contingent valuation. Both the limited database of nonuse ecological values suitable for a benefits transfer and concerns over the validity of contingent valuation estimates have hampered the inclusion of these values into CBAs.

This paper describes an alternative approach to estimating the nonuse damages associated with hypothetical natural resource injuries. The specific application is an analysis of the economic damages resulting from potential oil spills in the outer continental shelf (OCS) of the United States. Oil development in the OCS is managed by the Minerals Management Service (MMS), a bureau in the U.S. Department of the Interior. As part of its policy planning, MMS requires estimates of the potential external costs associated with OCS oil development.

We first discuss why contingent valuation estimates were rejected as the basis for our analysis. The paper then summarizes how natural resource agencies currently evaluate actual resource injuries using compensatory resource restoration. The methodology of habitat equivalency analysis (HEA) is described, including how it can be extended to evaluate potential resource injuries. A theoretical discussion describes the interpretation of HEA estimates in a welfare economics framework. The paper details the use of HEA for our specific application and presents results. Finally, the results are compared with other research to support our conclusion that HEA offers a viable option for estimating nonuse ecological values.

2. Contingent valuation and natural resource injuries

Under the Oil Pollution Act (OPA), natural resource trustees can recover nonuse losses so long as lost values can be reliably measured. Contingent valuation has traditionally been the only methodology available to estimate nonuse values. The National Oceanic and Atmospheric Administration (NOAA) convened an expert panel to explore the reliability of contingent valuation to assist in the final rulemaking process under OPA (NOAA, 1993). The panel concluded that contingent valuation estimates could provide useful information in the judicial determination of damages so long as the survey met certain scientific standards. The panel concluded that contingent valuation estimates tended to overstate actual losses and subsequent NOAA guidelines suggest that contingent valuation estimates be scaled downward by a factor of 2 in the absence of more reliable scaling information (NOAA, 1994).

The use of contingent valuation for estimating passive use values in natural resource damage cases remains a controversial issue (Binger et al., 1995; Portney, 1994; Renner, 1998). While some researchers have presented theoretical and empirical evidence supporting the reliability of contingent valuation estimates (Carson et al., 2001; Hanemann, 1994; Randall, 1993; Smith, 1996), others argue that contingent valuation estimates are not reliable (Diamond and Hausman, 1994; Stevens et al., 1991).

Even if one accepts the validity of contingent valuation, there exists a very limited database of contingent valuation estimates pertaining to the damages of coastal marine resources. Stevens et al. (1991) estimated the benefits of a salmon restoration program in Massachusetts, Silberman et al. (1992) valued the benefits of beach renourishment in New Jersey, and Le Goffe (1995) examined the benefits of preserving marine ecosystems in France. The only contingent valuation study that explicitly studies ecological injuries arising from oil spills is Carson et al. (1996). They asked Californians to assess the benefits of a program that over 10 years would prevent an unspecified number of oil spills, the deaths of 12,000 birds, and other ecological injuries.

Based on theoretical concerns over the validity of contingent valuation estimates and the limited availability of studies to use for a benefits transfer, we concluded that contingent valuation research could not provide a basis for evaluating the nonuse ecological damages of hypothetical oil spills.

3. Natural resource damage assessment and habitat equivalency analysis

The past decade has witnessed a significant shift in the objectives and procedures of natural resource damage assessment (NRDA). This transformation has refocused NRDA away from monetary damage estimates obtained using traditional welfare economics (Burlington, 2002). Instead, the current NRDA framework emphasizes public compensation through in-kind ecological restoration projects. The motives for the evolution of NRDA are both legal and practical (Flores and Thacher, 2002). First, federal statues concerning natural resource damages, such as the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and the OPA, mandate that public trustees spend compensatory funds only on “restoring, rehabilitating, replacing, or acquiring the equivalent of” natural resources (NOAA, 1994, 1996a,b). Compensatory funds cannot be directly distributed to those suffering a utility loss or disbursed to general municipal funds. The second motive is the difficulty of obtaining valid estimates of monetary damages, especially for nonuse values. While economic theory clearly supports the possibility of nonuse losses arising through natural resource injuries (Randall, 1993), the practical difficulty of presenting defensible estimates has caused the courts and public agencies to generally restrict the inclusion of nonuse values in NRDA cases (Robinson, 1996).

Natural resource agencies have clearly adopted compensatory resource restoration as a new paradigm in NRDA. A growing number of recent NRDA cases have obtained compensation from responsible parties in the form of natural resource restoration, including the 1996 *World Prodigy* oil spill in Rhode Island (NOAA, 1996a,b), the 1996 Chevron pipeline spill in Hawaii (Natural Resource Trustees for Pearl Harbor, 1999), and the 1997 Lake Barre spill in Louisiana (Louisiana Oil Spill Coordinators Office et al., 1999; Penn and Tomasi, 2002). Compensatory resource restoration determines the amount of some natural resource improvement required to compensate the public for losses associated with an environmental injury caused by a responsible party. While primary restoration aims to return the injured resource to baseline conditions prior to an accident, compensatory restoration is used to offset the interim loss of ecological services pending return to baseline conditions (Hoehn et al., 1996). The benefits of relying on compensatory restoration include cost-effectiveness, timely response and settlement, and greater cooperation among interested parties (Burlington, 2002). For example, all NRDAs conducted under the OPA have been settled without litigation using the compensatory restoration approach.

Habitat equivalency analysis (HEA) has been used to determine the appropriate scale of compensatory restoration in several recent NRDA cases. Resource compensation using HEA requires the following steps:

1. Choose one or more ecological metric(s) as indicators of the service provided by ecosystems;
2. Estimate the interim ecological service loss from the natural resource injury until the resource recovers to baseline conditions (with or without primary restoration);
3. Identify a range of compensatory restoration projects;
4. Choose one or more compensatory restoration projects that provide a present value of service gain equal to the present value of the service losses from the natural resource injury.

HEA is most straightforward when the compensatory projects provide services of the same type and quality as the injured resource. This allows a one-to-one scaling between the injury and the compensation using the same metric (NOAA, 2000). For example, the creation of 1000 acre-years of wetland services, measured as a discounted present value, compensates for an interim service loss of 1000 acre-years of wetland services so long as the created and injured wetlands are of similar type and quality. If identical compensatory projects are not available, then scaling ratios need to be determined. For example, created wetlands are often of lower quality than natural wetlands. HEA requires scaling ratios when the projects provide the same type of service, but of different quality. HEA also requires scaling ratios when out-of-kind projects are chosen for compensatory restoration (i.e., the compensatory projects provide a different type of ecological service than the injured resource).

HEA is appropriate for assessing damages to ecological services and is the preferred methodology when "the on-site uses are primarily ecological/biological and the off-site human uses are difficult to quantify" (Julius et al., 1995, p. 3). The Natural Resource Trustees for Pearl Harbor (1999) state that "HEA should be used in situations involving primarily the loss of ecological services with relatively little or no loss of direct human use" (p. 107). However, the presence of human use losses does not preclude the use of HEA. A number of NRDA cases have addressed ecological losses using HEA and human use losses separately based on market-based monetary techniques. If properly defined, use and nonuse damages do not overlap and these can be summed to obtain an estimate of total damages. For example, the 1996 *North Cape* oil spill NRDA (NOAA et al., 1999) calculated compensatory restoration requirements for injuries to both fauna, based on a HEA, and lost recreational fishing benefits, based on a benefits transfer of a travel cost analysis, summing these to obtain the total damage estimate.

While HEA has been used in a growing number of NRDA cases, it has not been used to evaluate potential natural resource injuries. As a practical matter, using HEA to evaluate potential injuries involves the same four basic procedural steps listed above but broader data requirements. The injuries from an actual oil spill or other hazardous release can be quantified through field observations and modeling. Hypothetical events involve a considerable degree of uncertainty regarding the probability, location, and magnitude of the release as well as the ecological and human impacts. Extension of HEA to estimate the potential damages from hypothetical injuries also raises a number of theoretical issues, which we now consider.

4. Theoretical issues

Under CERCLA and the OPA, the objective of natural resource damage cases is to make the public "whole." The public can be made whole with monetary compensation for natural resource injuries such that the gain in utility from compensation equals the utility loss from the injury (Jones and Pease, 1997). As stated above, funds collected in NRDA cases cannot be directly distributed to those who suffer utility losses and must, instead, be used for natural resource restoration projects.

This mandate raises an important issue regarding the scaling of compensation. Under the "value-to-cost" approach to scaling restoration projects, the costs of the restoration projects are scaled to the dollar estimate of utility losses (Chapman et al., 1998). If the utility gain provided to the public from these projects equals the lost interim utility until the resource recovers to baseline, then the public is fully compensated at the aggregate level. However, the benefits of compensatory projects cannot be guaranteed to equate to the lost utility. Consider a NRDA case where damages are estimated to be \$1 million. If a \$1 million settlement is used to fund various natural resource projects, these projects cannot be assured to provide an aggregate utility gain equal to \$1 million. The public may be over- or under-compensated depending on the benefits derived from the natural resource projects.

An alternative to the "value-to-cost" approach is the "value-to-value" method, which scales restoration projects so that the discounted utility from the natural resource projects equals the utility loss from the natural resource injury. Both values are estimated in dollar terms. The "value-to-value" method requires estimation of the expected utility of proposed natural resource projects, including both use and nonuse values. Estimating lost utility in dollar terms as a result of a natural resource injury is a difficult conceptual and practical task. Of course, these difficulties are compounded when analyzing a hypothetical project.

The disjunction between monetary measurement of damages and compensation via resource enhancement projects, along with the difficulties of obtaining valid monetary damage estimates, has led resource trustees to prefer compensatory scaling methods based on ecological services. Instead of measuring damages in monetary terms, "service-to-service" approaches such as HEA recognize that, as an approximation, the values humans place on natural resources are proportional to the ecological services these resources provide. Ecological habitats provide natural services including water quality maintenance, wildlife habitat, and flood control. By comparing the natural service gain from proposed compensatory restoration projects to ecological services lost as a result of an injury, compensatory projects can be scaled to fully compensate the public. Service-to-service approaches use an ecological metric, rather than dollars, to estimate damages. The metric can be biological, such as the number of birds killed or the lost primary productivity, or based on habitat characteristics, such as the loss of a given area of wetland habitat for a specific duration of time (e.g., an acre-year).

The utility loss an individual suffers as a result of a natural resource injury is measured in welfare economics as compensating variation—the monetary payment required to return the individual to his or her baseline level of utility. Using notation similar to Flores and Thacher (2002), compensating variation is defined as:

$$u_i^0\left(q_1^0, q_2^0 y_i\right) = u_i^1\left(q_1^0 - \Delta_1, q_2^0, y_i + CV_i\right) \tag{1}$$

where q is a measure of environmental quality or services. The injury reduces the reference level of services provided by a particular resource, q_1, by the quantity Δ_1 and the individual is fully compensated by a payment of CV_i. The total damage from the injury is the sum of compensating variation across all affected individuals. While Eq. (1) does not specifically incorporate a temporal dimension, one can easily extend the model to express compensating variation as a present value for injuries accruing until environmental services return to baseline levels. As discussed in Flores and Thacher (2002) and Unsworth and Bishop (1995), individuals can theoretically be compensated through resource enhancements instead of monetary payment:

$$u_i^0\left(q_1^0, q_2^0 y_i\right) = u_i^1\left(q_1^0 - \Delta_1, q_2^0 + \Delta_2, y_i\right) \tag{2}$$

where Δ_2 represents an enhancement to a resource, q_2, that may or may not be similar in type and quality to the resource injured.

Unsworth and Bishop (1995) note that the validity of this approach rests on the assumption that the costs of the resource enhancements do not significantly overstate or understate the actual (monetary) damages of the resource injury. They assert that "replacement costs are a poor cousin to theoretically correct welfare-based measures of economic damages" (p. 38). However, the theoretical advantages of welfare-based measures can be negated by the legal mandate that resource agencies must use collected funds for resource enhancement projects. Even if one is able to accurately measure and collect the "correct" economic damages of a natural resource injury, the utility value of the ultimate compensatory services provided to the public may exceed or fall short of the original damage.

The value-to-value approach described previously is the only method that guarantees the appropriate level of compensation from a welfare economics perspective. However, this approach is not used by resource agencies because of the difficulty in estimating the economic value of both the resource injury and the compensatory restoration projects. Conducting original economic research for each NRDA case is cost-prohibitive and the database of values that could be used in a benefits transfer is quite limited. Another issue relevant to ecological economics is that the value-to-value approach is anthropocentric—ensuring the maintenance of human welfare but not necessarily ensuring the ecological integrity of the environment.

The service-to-service approach, operationalized by HEA, seeks to maintain the discounted level of ecological services over time. The service-to-service approach can determine the "true" amount of compensation from a welfare economics perspective only under certain restrictive assumptions (Dunford et al., 2004). The most critical assumption is that humans derive utility from natural resources in proportion to the ecological services they provide. If so, the services from compensatory restoration projects should provide approximately the same level of utility as was lost from the natural resource injury.

If the present value of the utility provided by compensatory restoration projects is approximately equal to the utility loss from the resource injury, then in this respect the public is fully compensated. However, the monetary cost of restoration is a social cost-funds a responsible party pays towards restoration projects represent a loss to society. Thus, assuming appropriately scaled compensatory restoration, the cost of compensatory projects (including administrative and NRDA costs) equals the social costs of the injury.

This is the assumption operationalized in our analysis. We seek to estimate the compensatory restoration costs of hypothetical natural resource injuries as a measure of the social costs of such injuries. Despite the fact that HEA is a service-to-service approach, under these assumptions the costs of compensatory restoration also provide a measure of social costs.

5. Application background and scenarios

The Minerals Management Service manages the exploration and development of mineral resources on the Federal Outer Continental Shelf. The production of oil and natural gas from OCS lands is an important source of energy for the United States. OCS production accounts for about 27% of the natural gas and about 20% of the oil produced domestically. MMS divides the OCS lands into 26 Planning Areas: 15 in Alaska, 4 off the Pacific Coast, and 7 off the Atlantic and Gulf of Mexico Coasts. The vast majority of current OCS production occurs in the Gulf of Mexico.

Under the OCS Lands Act Amendments of 1978, MMS is directed to develop a series of 5-year leasing programs that consider the energy, economic, and environmental goals of the nation. As part of these programs, the costs and benefits of OCS activities must be considered in assessing the net economic effect of proposed OCS leases. The analysis described in this paper summarizes part of a larger research effort to estimate the external costs associated with OCS leases (Roach et al., 2001). These external costs include both the impacts from routine exploration and development activities and the costs associated with accidental releases of hazardous substances into the environment. The routine impacts considered in the larger effort include losses due to air pollution, reduction in private property values, infrastructure costs, reduction of commercial fishing profits, and impacts to subsistence resources in Alaska. The spill-related impacts include lost use values from beach recreation, commercial and recreational fishing losses, subsistence resource injuries, property value reductions, and lost ecological services. This paper focuses on the analysis of lost ecological services from potential oil spills as a result of OCS operations.

Separate estimates of potential ecological losses were required for each of the 26 MMS Planning Areas. Thus, the potential impacts of oil spills on a broad range of ecological habitats need to be determined. Oil spills related to MMS activity range significantly in magnitude—from minor releases of a few liters to spills of over 10,000 barrels (a barrel is approximately 159 l). Our analysis defines a broad range of spill scenarios that vary in location and spill size. MMS staff determined potential offshore spill locations for each OCS Planning Area. The number of potential spill locations in each Planning Area ranges from one to six (more locations in Planning Areas with greater activity) for a total of 50 locations. These locations vary from just a few miles offshore to over 100 miles offshore.

The majority of the volume of oil released into marine environments is a result of larger spills. U.S. Coast Guard data indicate that spills of less than 1000 gal (3785 l) account for only about 4% of the total volume of oil spilled into U.S. navigable waters since 1973 (USCG, 2003). Thus, we focus on spills of greater than 1000 gal (about 24 barrels). The MMS classifies a "large" spill as anything greater than 1000 barrels (MMS, 1999). Defining a "small" spill as a release between 1000 gal and 1000 barrels, we examined a MMS oil spill database to determine that the average small spill size from 1971 to mid-1999 was 111 barrels. The database was also examined to determine the average spill size for large spills (greater than 1000 barrels). For large spills from OCS pipelines or production platforms, the average spill size was 7000 barrels and for large spills from tankers the average was 25,500 barrels. For each of the 50 spill locations, our analysis considers six spill scenarios: two spill sizes (small and large) combined with three sources (pipelines, platforms, and tankers).

Spill occurrence rates need to be determined as our results were ultimately linked to OCS production levels. Using MMS data and the Coast Guard's Marine Casualty and Pollution Database, we determined historical spill rates for each of the six size/source combinations (see Table 1). Pooling the data in Table 1, we estimate that the production of one billion

Table 1
Oil spill magnitude and rates for spill scenarios

Variable		Spill source		
		Platforms	Pipelines	Tankers
Small spill	Spill rate[a]	4.23	12.43	1.14
	Average size[b]	111	111	111
Large spill	Spill rate[a]	0.45	1.32	1.21
	Average size[b]	7000	7000	25,500

[a] Numbers of spills of a given size per billion barrels produced/handled.
[b] Measured in barrels.

barrels of oil is expected to result in a total of about 21 spills greater than 1000 gal and a total release of about 45,000 barrels of oil into the marine environment, mostly from the likelihood a large tanker spill.

6. Scenario modeling

A total of 150 spill scenarios are analyzed based on 50 spill locations and three spill sizes (111, 7000, and 25,500 barrels). Each scenario required an estimate of the ecological service losses, in a format compatible with a generalized HEA. We used the Natural Resource Damage Assessment Model for Coastal and Marine Environments (NRDAM/CME, version 2.4) as a basis for all ecological service loss estimates. The NRDAM/CME is a publicly available computer program that models the physical, biological, and economic impacts of chemical releases into marine environments anywhere in the United States. The model requires input data on variables such as the spill location and magnitude, a wind time series, and the chemical spilled. The model produces a two-dimensional trajectory for the chemical over time. The model calculates ecological impacts based on an underlying database of the abundance of various fauna, the location of nine different coastal habitat types (saltmarsh, mud flats, sand beach, etc.), predominant marine currents and tides, and the toxicity of different chemicals. Ecological impacts are measured using three metrics: the number of wildlife killed of each affected species (including bird, marine mammal, and reptile species), the magnitude and duration of coastal oiling for each affected habitat, and the magnitude and duration of the oil slick.

Each of the 150 spill scenarios was modeled using the NRDAM/CME. We chose input variables to maximize ecological impacts. All spills, except for those in Alaska, are modeled to occur on May 1 given that biological impacts are likely to be greatest during the spring (French, 2000). All Alaska spills occur on August 1, again to maximize biological impacts. A wind time series was chosen from the historical record of offshore buoy measurements nearest to each spill location to maximize the frequency of onshore winds.

7. Generalized habitat equivalency analysis

Completed NRDA cases describing a HEA for a marine environment serve as a basis for generalizing the HEA approach to our application. HEA cases indicate a strong preference for in-kind restoration. Compensatory damages have been assessed for injury to habitats (e.g., the 1993 *Miss Beholden* grounding in Florida and the 1996 Chevron pipeline spill in Hawaii), injury to species (e.g., the 1986 *Apex Houston* spill in California and the 1989 *World Prodigy* spill in Rhode Island), or injury to both species and habitats (e.g., the 1997 Lake Barre spill in Louisiana and the 1993 Tampa Bay spill in Florida). Similar to the injury quantification in these NRDA cases, we measure injuries to wildlife as the number of animals killed for bird, reptile, and mammal species. Fish injuries are not specifically assessed as ecological impacts (injuries to fish are indirectly considered in the recreational and commercial fishing damages). We also include the injury to coastal habitats, as several NRDA cases included these separately from injuries to wildlife species. Several NRDA cases (e.g., the 1993 Chevron pipeline spill in Hawaii; the 1997 Lake Barre spill in Louisiana) have measured habitat injuries using the "acre-year" metric, the loss of ecological services from 1 acre for the duration of 1 year.

Both injury metrics are compatible with the NRDAM/CME output. The NRDAM/CME output provides estimates of the number of wildlife killed and coastal oiling, by habitat type, measured in square meter-days. The square meter-day metric is easily converted to acre-years. These outputs provided the basis for the "debit" quantities for our generalized HEA.

The NRDAM/CME output estimates the cost of required primary restoration activities. Legal decisions in NRDA cases have concluded that primary restoration should be undertaken unless it is technically infeasible or if the costs of primary restoration are "grossly disproportionate" to the benefits from reducing the interim losses until recovery (Mazzotta et al., 1994). The NRDAM/CME output indicated that primary restoration activities were not necessary for any of the spill scenarios we analyzed. The model output also indicated that natural recovery would occur for all scenarios within 1 year of the spill—thus, no discounting of future losses was necessary.

The next step was to determine the restoration actions and costs that could be used as "credits" to balance against the ecological injuries. The OPA guidance document on primary restoration (EG&G, 1996) is the most comprehensive assembly of data regarding ecological restoration activities, effectiveness, and cost. While the report focuses on primary restoration, it also discusses many possible compensatory restoration activities such as stocking, planting, and habitat construction. The NRDAM/CME report (French et al., 1997) provides additional information on restoration options. The U.S. Army Corps of Engineers has compiled a review of both Corps (Muncy et al., 1996) and non-Corps (Shreffler et al., 1995) restoration projects. NRDA cases were also reviewed for estimates of restoration actions and costs. We conducted a further literature review to collect additional data on restoration options and costs for coastal habitats (see Roach et al., 2001, for details).

Our literature review produced a total of 130 estimates of restoration actions with specific data on costs per acre (or costs per individual for wildlife). Of these, 50 applied to saltmarsh habitats and 34 to marine mammals (see Table 2). Few restoration cost estimates were available for the habitat types of rocky shore, gravel beach, and sand beach. The restoration cost estimates could vary significantly for a specific habitat. For example, the per-acre restoration costs for saltmarsh ranged from \$233 to \$450,000 per acre (in 1999 dollars). The broad range of costs suggests that one cannot choose a single representative estimate for the restoration of a specific ecological service or habitat. On the other hand, defining the upper- and lower-bound as the range of all available estimates could result in such a broad range as to be meaningless for policy analysis. Our approach was to define a "high" and "low" restoration cost estimate for each injury type that represents a likely range of costs that would arise in an actual NRDA (see Table 2). While sufficient data were available to determine a reasonable range of costs for several habitats, such as saltmarsh and macrophyte beds, limited data were available for other habitats. The available data suggested that restoration costs would be highest for saltmarsh and coral reefs—the two most ecologically productive and diverse habitats in our analysis. Thus, we determined that the restoration cost estimates for the other habitats should be less than the values for saltmarshes and coral reefs. As seen in Table 2, we also sought to maintain a relatively constant proportion between our "high" and "low" estimates—between about 5:1 and 7:1. In some cases, such as the gravel beach habitat, the cost estimates represent our best judgment based on data from other habitats.

Table 2
Final compensatory restoration cost estimates

Habitat/species type	Low estimate	High estimate	Number of cost estimates from literature review
Rocky shore	\$3000/acre	\$20,000/acre	1
Gravel beach	\$5000/acre	\$30,000/acre	0
Sand beach	\$7000/acre	\$40,000/acre	2
Mud flats	\$5000/acre	\$30,000/acre	1
Saltmarsh	\$10,000/acre	\$50,000/acre	50
Mangrove	\$3000/acre	\$20,000/acre	7
Macrophyte beds	\$4000/acre	\$30,000/acre	17
Coral reef	\$10,000/acre	\$50,000/acre	3
Mollusk reef	\$3000/acre	\$20,000/acre	3
Birds	\$500/individual	\$2500/individual	8
Mammals	\$13,000/individual	\$74,000/individual	34
Reptiles	\$2800/individual	\$9300/individual	4

An injury of 1-acre year of saltmarsh service does not imply that compensatory restoration of 1 acre of saltmarsh is required. First, compensatory restoration normally provides ecological service over a time span of more than 1 year. Thus, the service provided by compensatory restoration is calculated as a discounted present value. A discount rate of 3% is common in NRDA cases (NOAA, 2000) and is used in this application. Based on the analysis of Penn (1999) for the 1997 Lake Barre oil spill in Louisiana, we assume that restoration projects have a lifespan of 25 years and that the ecological service provided by compensatory restoration increases linearly over the first 3 years of the project as the habitat matures and then decreases linearly to zero over the remainder of the project life.

Another issue is that the ecological service provided by compensatory restoration may not be of the same quality as the injured resource. Restored or created habitats are typically of lesser quality than “natural” habitats. For example, restored wetlands normally do not achieve the same level of ecological function as natural wetlands (A.T. Kearney, Inc., 1991). Experiments by Zedler and Calloway (1999) indicate that constructed wetlands do not compare with natural wetlands in terms of soil organic matter, soil nitrogen, plant density, and other characteristics. There is also a chance that habitat restoration projects will fail. These factors imply that compensatory restoration should be provided at a ratio greater than one-to-one when compared to the injury. Compensation ratios for wetland mitigation banks range from 1.5:1 up to 10:1 (Environmental Law Institute, 1994). We assume a compensation ratio of 2:1—restored or created habitats at maturity provide half the ecological service of natural habitats.

Using a 2:1 compensation ratio, we assume that the service value from compensatory restoration starts at 0 and rises linearly over 3 years to 50% of the service value from a natural habitat. In other words, an acre of compensatory habitat will provide 0.5 acre-years of ecological service during its third year. The service value of compensatory restoration then decreases linearly from 50% of a natural habitat to 0 after 25 years. The service value for each year is then discounted to a present value using a 3% discount rate. These calculations imply that over the 25-year lifespan of a compensatory restoration project, 1 acre of compensatory habitat will provide 4.23 acre-years of ecological service in present value terms. Conversely, each acre-year of service injury to a natural habitat requires compensatory restoration of 0.24 acres of restored or created habitat. For example, assume a hypothetical spill oils 84 acres of saltmarsh and results in the loss of the marsh services for 3 months until natural recovery to baseline. The ecological injury in this example would be 21 acre-years of saltmarsh service. Using our assumptions regarding the value of compensatory restoration, the required compensation would be about 5 acres (21/4.23) of created or restored saltmarsh.

Restoration costs for wildlife deaths are calculated on a per-individual basis using the values in Table 2. No discounting or adjustment is required because restored individuals are assumed to serve as direct replacements for wildlife killed as a result of the spill.

This analysis allows us to determine the restoration requirements and costs for the injury estimates produced by the NRDAM/CME. Consider a simple example of HEA restoration calculations. Assume that a hypothetical oil spill injures both saltmarsh and sand beach habitats by the amounts indicated in Table 3. The injury produced by the NRDAM/CME in terms of square meter-days is first converted to acre-years of service loss. As described above, each acre of restored habitat provides a present value of 4.23 acre-years of service value. Thus, dividing the service loss (debit) measured in acre-years by 4.23 produces the number of acres of habitat that must be restored to cancel the debit. Once the required compensation is determined, the costs of restoration are obtained from Table 2. In the example in Table 3, compensation requires the restoration of 0.40 acres of saltmarsh and 8.64 acres of sand beach. The total restoration costs range from \$64,480 (low estimate) to \$365,600 (high estimate). This range represents the social costs of ecological injuries from the hypothetical spill.

Table 3
Example of generalized HEA compensatory restoration calculations

Variable	Saltmarsh habitat	Sand beach habitat
Debit calculations		
NRDAM/CME service loss (square meter-days)	$2.50*10^6$	$5.40*10^7$
NRDAM/CME service loss (acre-years)	1.69	36.56
Credit calculations		
Credit per restored acre (acre-years)	4.23	4.23
Required compensatory restoration (acres)	0.40	8.64
Cost (low estimate)	\$4000	\$60,480
Cost (high estimate)	\$20,000	\$345,600

8. Scenario results

Using the above methodology, compensatory restoration costs were calculated for all 150 spill scenarios. The external costs of the other spill-related impacts (beach recreation, commercial and recreational fishing, subsistence resources, property value losses) were also estimated for each scenario as well as the NRDA administrative costs (see Roach et al., 2001 for details). The scenario results indicate that spill damages vary significantly by spill location. As might be expected, economic damages are much larger for those spills that reach the shore. Expected damages for specific spill scenarios range from a low of $11,000 up to $215 million.

The damage estimates for each Planning Area were converted to a per-barrel-spilled basis. Table 4 presents the results for each spill size and each type of damage (weighing all OCS Planning Areas equally). The marginal damages per barrel spilled decrease with increasing spill size. Note that the economic damages to ecological services (injuries to coastal habitats and wildlife species) comprise the vast majority of all damages from the spill scenarios. Most spill scenarios cause comparatively little damage to human use values, especially in the sparsely populated Alaskan Planning Areas.

Considering both the low and high estimates, the range of damages per barrel spilled across individual Planning Areas is $27 to $14,418. Averaged across all Planning Areas, the average damage per barrel spilled is $982 using the low damage estimate assumptions and $4825 per barrel spilled based on the high damage assumptions.

Given that we are analyzing hypothetical scenarios, one cannot fully assess the validity of our damage estimates. However, the results appear reasonable when compared to other research. Sirkar et al. (1997), using an earlier version of the NRDAM/CME, estimated the damages from tanker spills as a function of spill size in a CBA of different tanker designs. For a spill the magnitude of our large tanker spill scenario, Sirkar et al. estimate NRDA damages of approximately $800 per barrel. As seen in Table 4, our damage estimates per barrel spilled for a large tanker spill are $571 using the low estimates and $2825 using the high estimates.

Reviewing data from numerous government agencies, Helton and Penn (1999) present NRDA values for 30 spills since 1984, including spills into freshwater environments and spills of chemicals other than oil. The NRDA amounts in these cases were generally not based on formal HEAs. The NRDA damages per barrel spilled range from $3 up to $15,759 with an average of $1745 per barrel spilled. These values are very comparable to our estimates, which range from $27 to $14,418 and average $2904 (considering both the high and low estimates). The per-barrel damages

Table 4
Average economic damages per barrel spilled from spill scenarios (1999 dollars)

Damage category	Small spill (111 barrels)		Large pipeline/platform spill (7000 barrels)		Large tanker spill (25,500 barrels)	
	Low estimates	High estimates	Low estimates	High estimates	Low estimates	High estimates
Coastal habitats	146	803	43	235	23	126
Wildlife deaths	4630	23,259	1059	5306	480	2404
Beach recreation	35	165	3	12	1	5
Recreational fishing	146	758	19	96	9	45
Commercial fishing[a]	236	236	19	19	10	10
Subsistence activities	2	7	<1	<1	<1	<1
Property value losses[a]	59	59	3	3	1	1
NRDA administrative	473	2276	103	510	47	233
Total	5727	27,564	1249	6182	571	2825
Percent of total damages from ecological injuries	81%	84%	85%	86%	84%	85%

[a] A "low" and "high" range was not calculated for commercial fishing and property values losses as the damages for these categories were based on specific values. For example, commercial fishing damages were calculated as the lost catch based on the NRDAM/CME output multiplied by average market prices.

for crude oil spills in Helton and Penn tend to be larger than the damages for non-crude spills. The average damage per barrel spilled for the crude oil spills in their analysis is $2714, which is very similar to our average of $2904 per barrel spilled.

Three NRDA reports not considered by Helton and Penn were identified, using the NOAA Damage Assessment and Restoration Program Internet site (http://www.darp.noaa.gov/), for which NRDA damages per barrel spilled could be calculated. For the 1990 Apex Barges spill, damages were $76 per barrel spilled. For the 1990 Apex Galveston spill, damages were $102 per barrel spilled. For the 1996 *North Cape* spill, damages were $1400 per barrel spilled. These values fall within the range of our damage estimates per barrel spilled.

9. Conclusion

Our analysis demonstrates that ecological service-to-service techniques represent a reasonable theoretical and practical alternative to traditional monetary valuation to assess nonuse losses. Habitat equivalency analysis has quickly become the dominant paradigm in NRDA cases. Our research illustrates that HEA can be generalized as a policy planning tool for natural resource injuries. While our application focuses on the impacts of offshore oil development, HEA could be applicable as a policy planning tool to a range of additional natural resource issues. For example, the damages from global climate change to ecological habitats and wildlife could also be assessed using HEA if sufficient data were available on the potential probability and magnitude of injuries.

HEA manifests the strong sustainability concept in that it seeks to maintain the provisioning of ecological services over time. HEA limits substitutability by requiring compensatory ecological restoration that, to the extent feasible, correlates with the ecological injury. The difference in ecological service values between a restored and a natural habitat can be addressed through the use of compensation ratios. While maintenance of an overall level of ecological services may satisfy some ecologists, the more important issue for economists (especially mainstream economists) is likely to be whether the service-to-service approach concurrently maintains human utility levels. An important issue for future research is to estimate the economic value of services provided by compensatory restoration and compare these values to the economic damages from natural resource injuries. This implies that the service-to-service technique does not obviate the need to estimate non-market values in monetary terms (Flores and Thacher, 2002; Ofiara, 2002). If the public is not being fully compensated (or over-compensated) from the new NRDA paradigm, then revision of the process may be warranted. As Dunford et al. (2004) note, the potential for HEA to produce reasonable estimates of compensation rests upon the validity of its underlying assumptions. Further analysis regarding issues such as the scaling of services across ecological habitats and appropriate compensation ratios is clearly desirable.

Using HEA as a planning tool also raises the issue of the validity of transfers and generalizations across habitats and geographic regions. Our analysis assumes that restoration costs, compensation ratios, and service flows are constant across Planning Areas. Future research by restoration ecologists and analyses of actual NRDA cases can expand the database we used for this study and possibly allow for greater analysis precision.

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