NATURAL RESOURCE DAMAGES VALUATION: ARTHUR KILL OIL SPILL

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ABSTRACT

The benefits transfer methodology is often used in regulatory settings. The relatively modest time and data requirements are advantageous, but those advantages must be weighed against the imprecision of the estimate. An important area for further evaluation is whether the transfer methodology can be used effectively in a natural resource damage assessment (NRDA). Because litigation is a possibility, the role of the transfer methodology needs to be carefully assessed for use in NRDAs. This paper discusses the Arthur Kill oil spill as a case study for such an evaluation.

The benefits transfer methodology is often used in regulatory settings because regulators find the relatively modest time and data requirements attractive. Although the transfer methodology allows a comparison of costs and benefits, it also has disadvantages. The primary disadvantage is the imprecision of the estimate; that imprecision becomes a major issue when the value of the estimate plays a major role in decision making. An important area for further evaluation is the role of the transfer methodology in natural resource damage assessments (NRDAs). NRDAs are undertaken when state or federal trustees file a legal suit to recover damages to natural resources caused by accidents such as oil spills. The possibility of litigation in such a setting creates some unique concerns.

This paper discusses the Arthur Kill oil spill as a case study for evaluating the transfer technique for NRDAs. It summarizes the discussion and conclusions reached by case study members at the 1992 AERE Workshop. The paper is organized as follows. After describing the background of the spill, we describe the transfers in an NRDA context. We follow with a discussion of data and methodology issues. Finally, we propose a research agenda to address key issues remaining in the evaluation of the transfer methodology in an NRDA context.

BACKGROUND OF THE ARTHUR KILL OIL SPILL

The rupture of an Exxon underwater pipeline an January 1, 1990, resulted in the release of 567,000 gallons of No. 2 fuel oil (not as light as gasoline but not as heavy as No. 6 fuel oil)

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into the Arthur Kill. The leak occurred from a 5-foot gash in the 12-inch pipeline that connects the Bayway Refinery at Linden, New Jersey, to the Bayonne Plant in Bayonne, New Jersey. The leak site is just south of the Goethals Bridge. The spill occurred near the New Jersey coast, but tides and winds moved the oil to the three islands in the Kill and the Staten Island coastline. The Coast Guard considered the spill to be “major” because it involved the release of more than 10,000 gallons (42 gallons = 1 barrel).

The clean-up crews recovered approximately 141,000 of the 567,000 gallons of oil. About 50 percent of the oil evaporated. Clean-up crews completed the clean-up on March 15, 1990. The Bird Rescue phase of the clean-up resulted in the treatment of 150 birds, of which 110 survived (Exxon internal company documents).

DESCRIPTION OF THE ARTHUR KILL AREA

Figure 1 provides a geographic description of the area, including the affected regions of the Kill and the extent of oiling. The Arthur Kill is a waterway located between Staten Island, New York, and the New Jersey coastline near the Newark airport. The Arthur Kill is approximately 15 miles long and almost 2 miles wide; it opens into the Raritan Bay at the south end and the Kill van Kull and Newark Bay at the north end. It provides access to New York Harbor, Raritan Bay, Lower Bay, Jamaica Bay, Newark Bay, the Hudson and East Rivers, and the Atlantic Ocean from the New Jersey coast of Newark, Linden, and Elizabeth.

The entire Arthur Kill is surrounded by salt marshes and salt- and freshwater creeks, which support harbor bird habitats on the three islands within the Kill: Prall’s Island, the Isle of Meadows, and Shooter’s Island. The area around the Arthur Kill is circumscribed with salt marshes and estuaries which serve as nurseries for well over 145 different species of fish and birds. Although the Arthur Kill is an intensely developed industrial area, several species of wading birds remain in the area in large numbers. Paradoxically the industrial activity actually provides relative seclusion from humans, making this an ideal breeding ground (The Trust for Public Land and New York City Audubon Society, 1990). Creeks run from the Kill onto Staten Island and New Jersey, creating wetlands areas that are essential feeding areas for the birds. The actual nesting sites for the bird species are located on the three rookery islands (The Trust for Public Land and New York City Audubon Society, 1990).

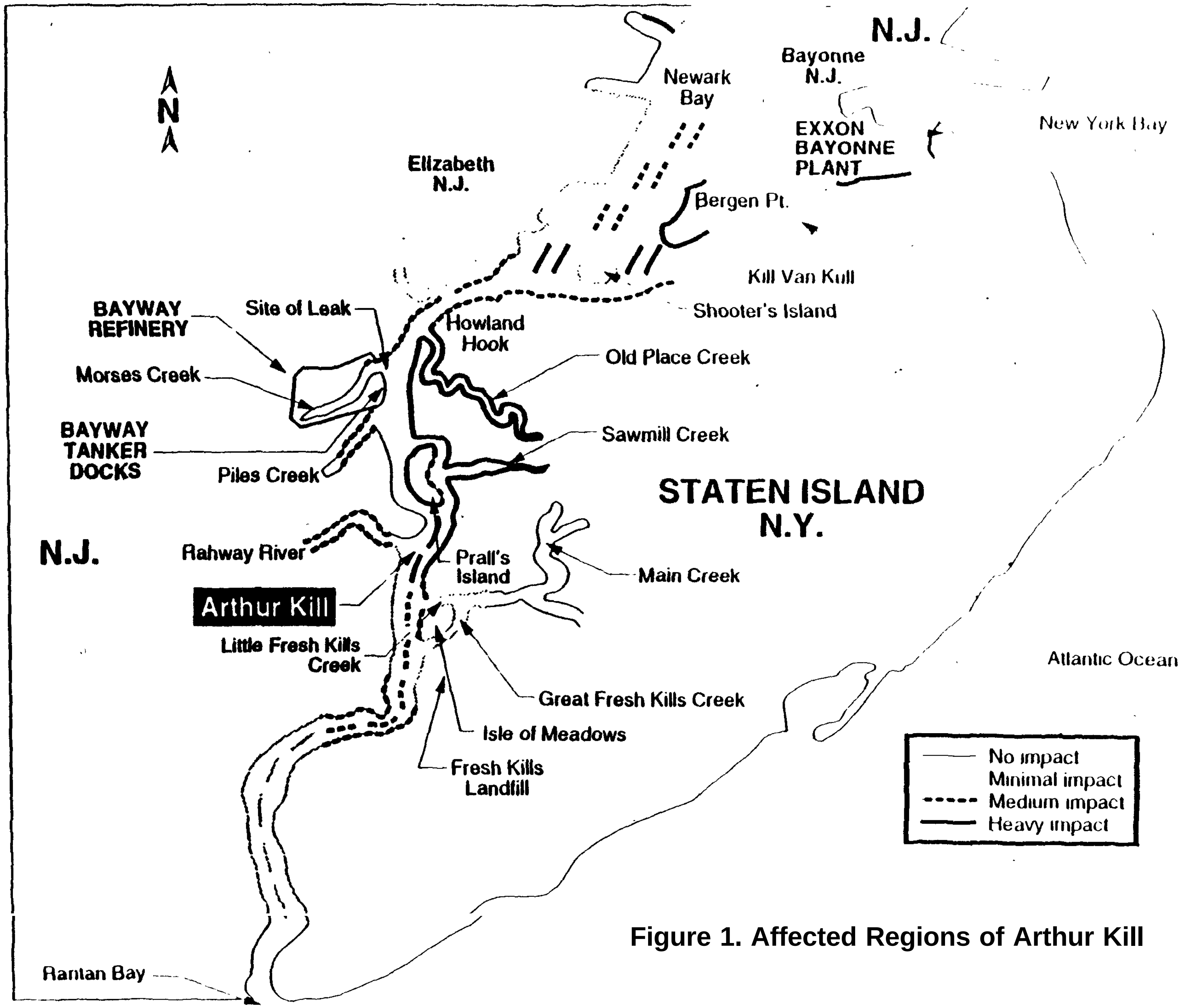

Figure 1. Affected Regions of Arthur Kill

The Arthur Kill and Kill van Kull are both bordered by a variety of industries, predominantly chemical manufacturing and oil refining. As a result of the intense industrialization and the proximity of the New York City and the Linden, New Jersey, landfills, the area is vulnerable to industrial and municipal pollution. The New York City landfill, Fresh Kills, is directly adjacent to the Kill and is the largest in the United States, towering about 500 feet above the Arthur Kill. City reports indicate that the water quality in the Arthur Kill is the poorest in the New York Harbor area as a result of the heavy industry and the presence of Fresh Kills and a smaller landfill nearby (Urbont, 1990).

The Arthur Kill connects other large bodies of water, as does the Kill van Kull. The Arthur Kill connects Newark Bay to Raritan Bay and the Atlantic Ocean. The Kill van Kull connects the Upper New York Bay to Newark Bay; the Upper Bay is adjacent to the Hudson and East Rivers, the Lower Bay, and the Atlantic Ocean. The waterways are narrow and are used frequently by commercial shipping tankers. Because this area is intensely industrialized and the water quality is fair to poor, recreation activities in the Kill itself are limited or nonexistent.

However, for the residents of New Jersey and to some extent New York, the Arthur Kill and Kill van Kull are important as access waterways to reach the adjacent areas where recreation opportunities are more abundant. They serve as an access for fishing in Raritan Bay, Lower Bay, Newark Bay, and the Hudson and East Rivers and for boating access in the same areas with the addition of New York Harbor. A great deal of water-based and land-based recreation takes place in and around Raritan Bay (south of the Arthur Kill), particularly near the New Jersey portion of the Gateway National Recreation Area. Likewise, the Atlantic Ocean side of Staten Island offers recreational activities at the Great Kills Park (eastern shore) and the other portion of the Gateway National Recreation Area. The Jamaica Bay Wildlife Refuge and several popular beaches are across the Lower Bay and adjacent to Brooklyn and Long Island. Because these recreation areas are so close to New Jersey, the Arthur Kill and Kill van Kull offer inexpensive, convenient access to popular recreation sites in the greater New York area.

Four marinas are located either on the Kill or within a mile of the Arthur Kill or Kill van Kull. One of the marinas offers a public boat ramp while the others predominantly provide slip storage for moored boats. In addition, six city and county parks are within the area. Park officials indicated that these parks were used for picnicking, bird watching, and other activities such as softball.

In addition to use services, the Arthur Kill area contains wetlands that may provide nonuse services. The wetlands system in the Arthur Kill area covers approximately 400 acres of freshwater and saltwater tidal marshes and creeks. The areas where potential effects may be found are along the Kill between Bridge Creek (north of Goethals Bridge) and the Isle of Meadows (at the mouth of Fresh Kills). This area covers approximately 127 acres of wetlands (B-Laing, 1900) and supports a variety of wading and seabird species as well as several hundred invertebrate species. The freshwater marshes support an additional 20 to 30 species of invertebrates and vertebrates suitable as food for the birds in the area. The area directly contributing to habitat functions covers approximately 25 to 40 acres (the sum of the acreage used for feeding and nesting).

Biologists assess wetlands in terms of their functions using a qualitative method of evaluation called WET, which stands for Wetlands Evaluation Techniques. WET analyzes the wetlands area in terms of social significance, effectiveness, and opportunity. The Exxon technical team conducted a WET analysis on the Arthur Kill region. Its results are cumulative for the many oil spills that occurred in the region in a short time period, implying that the effects are likely to be greater than just those from the Exxon spill. Conclusions about the functions of the Arthur Kill wetlands include the following:

- The Arthur Kill wetlands have a *limited* potential to nourish plant and animal life in the area as well as provide eutrophic effects downstream. This function is limited because the commercial and recreational traffic in the area is significant.
- These wetlands serve as a filtering system by trapping sediment, pathogens, and toxic substances and removing them from water transport. Again, the extent of this function is uncertain because of the water traffic.
- These wetlands provide an important educational and research function. In particular, the Harbor Herons Project has served to educate the public. The fact that these

interesting and beautiful creatures are increasingly populating two islands in a metropolitan area of more than 15 million provides a unique ecological case study. In 1986, volunteers engaged in heron projects involving the building of heronries and presenting information about the area and its species. This project generated a great deal of media coverage (Parsons, 1986).

- The last function the Arthur Kill wetlands provide is erosion control protection for the area. The region provides moderate erosion control, particularly along the shorelines of the wetlands where peat sediment is stabilized by the intertidal marshes, which contributes to a stable shoreline and deters erosion of the mainland (Winfield, 1990).

In summary, this natural resource setting provides the backdrop for a case study using the benefits transfer methodology in an NRDA context. The setting enables researchers to evaluate both use and nonuse natural resource services. Use values are the values associated with natural resource services where physical and/or visual contact between people and the natural resource occur. Nonuse values do not require contact; rather these services are the result of a resource providing well-being to people or other resources simply by existing.

Both National Oceanic and Atmospheric Administration (NOAA) and Exxon prepared damage estimates using the transfer methodology. The parties were able to reach a negotiated settlement based on these estimates. Because the estimates prepared by the NOAA have not been made public, this paper relies on the estimates prepared by Exxon's experts, which have been made public (Desvousges and Milliken, 1991).

TRANSFER STUDIES IN AN NRDA CONTEXT

The Arthur Kill oil spill is typical of many NRDA cases. The size and/or location of such spills often make a full-blown damage assessment inefficient because the assessment itself could cost more than the damage. In these instances using the transfer methodology to estimate the damage is more efficient. Benefits transfer methodology also can provide a useful screening devise for targeting assessments that will require more detailed Type B assessments.[1]

During the AERE workshop, participants discussed using transfer methodology in NRDAs. The level of comfort among participants in using the transfer methodology (or willingness to use the transfer methodology) depended on the status of the assessment and the amount of probable scrutiny it will receive. As part of this discussion, the participants discussed a continuum of NRDAs: on the left side are initial screening assessments and on the right side is

[1] The Type A model is a simple process that uses a standard computer model and requires minimal input data. It is most useful for small, short-duration marine and coastal spills. The Type B assessment applies to all other releases in coastal and marine environments and releases involving freshwater and land resources, including plants and animals. Type B assessments are more complex and comprehensive in which damages are determined through a three-step process: injury determination, quantification of service effects, and damage determination.

a full-blown study to support litigation (see Figure 2). The continuum depicts the role of the assessment. As the NRDA progresses from an initial assessment to negotiated settlement to litigation, scrutiny increases. Thus, the imprecision associated with using the transfer methodology may be more of an issue when litigation is bending.

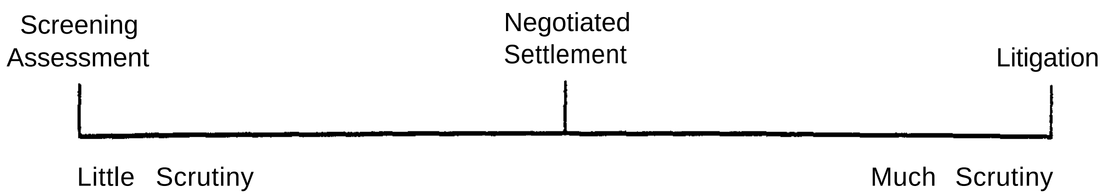

Figure 2. Continuum of Valuation Scrutiny in an NRDA Context

When the level of scrutiny is relatively low, the willingness to use the transfer methodology is high. None of our group members expressed any hesitation about using transfers for an initial NRDA or for a negotiated settlement. However, most members were reluctant to adopt the transfer methodology when litigation is involved. Because the level of scrutiny is much higher in a litigation context, most of our group members thought that the margin of error inherent in a transfer study was not defensible.

Finally, the transfer methodology can also be used in establishing and implementing NRDA policy. For example, the Type A[2] assessment used by NOAA to estimate the damage caused by certain types of oil spills is a transfer model. The budget and time constraints for NRDA policy making are similar to the types of constraints that make transfer methodology attractive to litigants. None of our group members expressed serious concerns about using transfer studies in a policy-related context.

DATA AND METHODOLOGY ISSUES

Like any transfer study, data and methodological issues need to be resolved for NRDA transfers to be effective. In our discussions, we identified three types of data and methodological problems likely to arise in NRDA transfer studies: development of the quantity data (the number

[2] See footnote 1.

of damaged resources or services), valuation of the interrupted or eliminated services, and the valuation methodology.

The development of the quantity data to be used in an NRDA transfer study often proves to be challenging. Recall that, in a transfer study, the value or price of the affected resource or service is transferred from other studies. However, the researcher must determine the quantity of affected resources or services before applying the transferred values. Determining the quantities for an NRDA typically requires judgment because the quantities of resources or services are not observable, or if observable, are not readily available. If historical data are available, they may provide some useful guidance, but judgment may still be necessary.

In the Arthur Kill oil spill, the Exxon estimate addressed three types of interrupted or lost services: fishing and boating access, near-water recreation (park use), and wetlands services. The access data used in Exxon's estimate were based on interviews with local marina operators. The number of marina slips and an estimate of typical occupancy during the winter months were combined to estimate the number of affected boats in marina slips.

The park-use data were also based on interviews with key informants (park officials in this case). They estimated park use in terms of the number of visitors during the off-season. Finally, biologists estimated the number of acres of affected wetlands based on their field assessment.

Relying on key informants to develop the quantity estimates is not unusual in an NRDA. In many cases, no better source of data is available. However, using key informants may introduce moral hazard into the picture because they may have a vested interest in the outcome of the damage estimates. Key informants may realize that the interviewer is somehow associated with the recent spill, and the informant may provide biased estimates.[3] When relying on key informants for the quantity data, researchers should use their best judgment and be aware of the possibility of moral hazard.

Biologists or other types of scientists often provide other types of data, such as quantity estimates. In many instances, estimates by scientists are the best source of the necessary data. Our group expressed some concerns about relying on this type of data also. Scientists often approach issues differently from economists, thus producing data that are not useful to economists. Our group discussion indicated that economists and scientists should coordinate their future efforts better than they have historically.

[3] Although commercial enterprises cannot bring an NRDA suit, they may make commercial claims. The data they provide may be biased to secure a larger commercial claim.

The second type of data issue we discussed was valuation issues. In a transfer study, the quality of the estimate depends partly on the availability and quality of the original studies and their suitability to transfer. These concerns are not unique to NRDA transfer studies. Adopting the transfer methodology means that the researcher adopts the values from the original study and any inherent weaknesses in them.

In addition to methodological issues, we confront issues of “sameness” as well. The NRDA estimate based on a transferred value has more credibility when the affected service is very similar to the service on which the transfer value is based. For example, an NRDA estimate for cold-water fishing in the Northeast may not be well represented by a value for warm-water fishing in California. Seasonality is an important consideration for many recreation estimates. Most recreation studies are based on the “high season,” the season when that particular recreation activity is at its peak. Ignoring the seasonality issue in a transfer study can result in error in the NRDA estimate.

As part of the discussion, we informally polled our group members on their assessment of the adequacy of existing studies for transferring use values. We asked our group members to rate the existing studies on a scale of 1 to 5, with 1 being inadequate and 5 very adequate. This assessment included the number of studies, the quality of those studies (inclusion of substitute sites, assumptions, parameters), and the “transferability" of the studies. Table 1 shows the general adequacy ratings, although the adequacy of available studies will vary in particular cases (e.g., locations, season, activity). For many types of use services, the group consensus was that existing studies are generally not adequate for transfer. We concluded that existing studies on big-time sport fishing and big-game hunting are more adequate for transfer purposes than the other use categories considered. Existing studies on other uses such as swimming and wildlife viewing did not receive a favorable rating in terms of adequacy.

Data issues for nonuse values ate particularly controversial. Even in a full-blown analysis, nonuse values are extremely difficult to estimate. Economists have used contingent valuation (CV) to estimate nonuse values, and disagreement exists about its validity for this use. The difficulty of the situation is amplified in a transfer study.

TABLE 1. GROUP ASSESSMENT OF THE ADEQUACY OF EXISTING STUDIES FOR TRANSFERRING USE VALUES

Use Category	**Adequacy**[a]
Fishing	
Big time	4
Small time	3
Boating	
Motorized	1
Nonmotorized	2
Swimming	1
Beach Use	1
Shoreline Use	1
Wildlife Viewing	1
Hunting	
Big game	4
Waterfowl	3

[a] **1=inadequate;** 5=very adequate.

We specifically evaluated the adequacy of studies for wetland values. In the Arthur Kill study, the biologists determined that the wetlands in the Kill area were only serving some of their intended functions. A study that focused on this particular subset of functions in the same geographic area did not exist. The studies that do exist do an incomplete job of valuing wetlands, even in general terms. Table 2 summarizes the available wetlands studies.

The methodological issues we discussed focused on the unit of valuation. Use services studies have four possible choices for the unit of valuation. The first is the unit-day value, where, for example, the value of a fishing day or a boating day is transferred from a study to the NRDA site. Although this approach has the advantage of simplicity, the differences between the sites that may influence the demand for services are essentially ignored.

The second approach uses a valuation equation. In this type of transfer, the coefficients from an existing study are applied to the means (or representative values) of the same variables for the NRDA site. This approach offers an improvement over the unit-day value, but it is

TABLE 2. VALUATION STUDIES OF WETLANDS

Authors	Title of Study/Article	Scope of Study/Issues Addressed	Analysis Used	Conclusions
James G. Gosselink Eugene P. Odum R. M. Pope	"The Value of the Tidal Marsh"	Monetary evaluation of natural tidal marshes in Louisiana using energy methodology analysis.	Energy Analysis	The author concludes that by-product production yields lower per-acre values than more intensive uses that preserve the natural functions of systems. The author computed ecological life-support values based on gross primary productivity (in energy terms) of the natural marsh, using a conversion ratio from energy lo dollars based on the ratio of Gross National Product to National Energy Consumption The resulting wetlands values are $2,500-$4,000 per acre per year.
Leonard A. Shabman Sandra S. Batie Carl C. Mabbs-Zeno	"The Economics of Wetlands Preservation in Virginia"	Development of values of wetlands using value per la in Virginia Beach and value per acre of recreational home subdivisions to analyze permit decisions. Coastal states have frequently tried to establish legislation to diminish the rate at which coastal wetlands are being reclaimed for different forms of development. This article discusses the ecosystem services of a wetlands by developing a model of structures and functions. Then, it examines the process of development and permitting for development is examined.	Market Valuation	Two case examples are examined in terms of the development values of wetlands. In developing Virginia's wetlands, residential home development and recreation home development comprise the two main pressures. Two valuation studies provide insight into this type of development using hedonic price equation to estimate regressed land sale prices on a set of explanatory variables representing individual land parcel characteristics, including measures of water access and waterfront location created from filled wetlands.
John C. Bergstrom John R. Stoll John P. Titre Vernon L. Wright	"Economic Value of Wetlands-Based Recreation"	Valuation study to quantify the outdoor recreational value of wetlands. An empirical study was conducted on current recreational uses of a coastal wetlands area in Louisiana.	Willingness to Pay Expenditure Analysis	The study yielded $17.10 as an annual per-acre value of wetlands. The results suggest suggest economic impacts and net economic benefits associated with wetlands recreation.

(continued)

TABLE 2. VALUATION STUDIES OF WETLANDS (CONTINUED)

Authors	Title of Study/Article	Scope of Study/Issues Addressed	Analysis Used	Conclusions
Robert Costanza Stephen C. Farber Judith Maxwell	"Valuation and Management of Wetland Ecosystems"	Study of wetlands values in coastal Louisiana that employed willingness-to-pay and energy-analysis-based methodologies. This article discusses the fundamental theoretical and practical problems underlying resource valuation. It summarizes the methods and findings for the Louisiana wetlands study.	Willingness to Pay Energy Analysis	The authors were able to bracket a range of values. They estimated that $194.32 to $512.00 is the per-acre value of wetlands per year. The low end of the range is based on the willingness-to-pay approach, and the upper end is based on the energy-analysis approach. The largest value provided based on the energy-analysis approach was $848 per acre per year.
Robert Costanza Stephen C. Farber	"The Economic Value of Wetlands Systems"	This study uses a willingness-to-pay and an energy-analysis method of establishing the social value of a wetlands system. The economic approach considers the commercial, recreational, and storm protection value of wetlands. The energy analysis evaluates the energy processed by the wetlands system in south Louisiana.	Willingness to Pay Energy Analysis	The economic value of willingness to pay for an acre of wetlands by type tanged from $0.44 to $37.46 for the annual value per acre. The energy analysis evaluation considers the total amount of energy captured by natural ecosystems as an estimate of their potential to do useful work for society. This method provides a comprehensive upper hound on the economic value of the system's products. Using the transformation of salt marsh to open water, the annual value of loss of wetlands ranges from $509 to $847 per acre per year.

frequently difficult to find an appropriate equation to transfer and the comparable data for the NRDA site.

The third approach is a generalized model from which values can be transferred. Such a model requires much more information that the previous two approaches, but it offers the advantage of better estimating the site-specific value. The group members discussed the possibility of adapting the Random Utility Model (RUM) for transfer.

A final option for valuation is the meta-analysis approach. This approach compiles all available values and their influences and produces a value that accounts for the many possible influences. Like the generalized model above, the data requirements are extensive. (For nonuse values, whether such an analysis can be performed given the currently available studies is unclear.)

The methodology adopted in an NRDA transfer study depends in part on the timing, the funding, and the available data. Our group discussion indicated that we would like to see a movement toward using the generalized model.

RESEARCH AGENDA

Our group discussion revealed that much research still needs to be done on use and nonuse values for NRDA transfer purposes. We focused on three primary research items: the design and undertaking of a “grand” study, more and better original studies, and a technique to generalize RUMs.

The first research agenda item (deemed most important by the group) was the design of the grand study. Such a study would encompass all types of services and the influences on the demand for these services. The study would be suitable for transfer purposes and would be linked to ecological models.

The second research item is the need for more and better quality original studies. Our group thought more studies on use values would be helpful, particularly on those types of values for which few studies exist, such as swimming and boating. But more important are studies on nonuse values. Such research should address fundamental issues associated with credible valuation procedures. Consensus on transferring nonuse values depends on consensus on estimating credible nonuse values. The group concluded that good studies on wetlands and seabirds would go far in filling our needs for nonuse estimates. New studies undertaken should be designed with transfer in mind.

Finally, we decided that our discipline should take steps to generalize RUM models for use in transfer. The goal of this research would be to evaluate how a RUM could be used in the transfer process. For example, would it be possible to design a large-scale data collection, such as a multistate region, that would support a general RUM model? Alternatively, another strategy might be to divide the collected data into subsets that could be used to estimate a RUM for a specific set of sites relevant for the transfer problem. Finally, the group agreed that better data are essential for using RUMs in a transfer setting.

This agenda is ambitious and requires funding. Sponsors of new studies have their own specific needs, and those needs may not correspond with transfer study needs. This last point may be particularly true of NRDA litigation situations. Sponsors of any such study have their own timetable and agenda and may not be willing to subsidize the purely research components of a study.

Finally, our group concluded that, as a discipline, we need to change our attitudes about replication. Such studies would be extremely helpful for transfer purposes, but traditionally such studies are not publishable. Consequently, researchers do not undertake replicative studies, or if they do, they are not published and generally not readily available to other researchers. However, we discussed the need to consider using experimental designs to evaluate the validity and reliability of the previous study. Research progress from simple replications would be far less informative.

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RECREATIONAL FISHING VALUATION: APPLICATION OF THE TYPE A MODEL

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ABSTRACT

The Type A model is the single largest benefits transfer model for natural resource damage assessment and the only one that has regulatory status for litigation under CERCLA and the Clean Water Act. In this case study, we focus on the Type A model procedures for valuing losses in recreational services due to fish kills and fishery closures resulting from an oil or chemical spill. In addition we discuss how to value recreational fishery injuries.

The natural resource damage assessment model for coastal and marine environments, the "Type A model," is the single largest benefits transfer model for natural resource damage assessment and the only one that has regulatory status for litigation under CERCLA and the Clean Water Act The model provides a simplified assessment procedure for short-term releases of oil and hazardous substances. It represents a low-cost alternative to Type B damage assessments, which may require detailed field observations and extensive collection and analysis of chemical, biological, and behavioral data.

The first-generation Type A model, under review here, was promulgated under rule-making by the U.S. Department of Interior (DOI) in 1987. It covers the coastal and marine environment of the U.S. DOI is required to revise the model every two years; this year, the agency intends to propose a new Great Lakes version, as well as a substantially revised coastal and marine version of the model.

In this case study, we focus on the Type A model procedures for valuing losses in recreational services due to fish kills and fishery closures resulting from an oil or chemical spill. The Type A model incorporates the data and algorithms to calculate fishery injuries, measured as the reduction in (fish stocks and) recreational fishery catch

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by weight due to direct kills, recruitment losses, food web effects, and closures. The problem posed in this case study is how to value such recreational fishery injuries.

BACKGROUND: TYPE A MODEL FOR COASTAL AND MARINE ENVIRONMENTS

The model relies on computer modeling to predict the fates and effects of spills and value the injuries. The essence of the operation is contained in three modules: physical fates submodel, biological effects submodel, and economic damages submodel (see Figure 1).

The physical fates module models the path of contamination as it disperses, determining the concentration of the spilled substance over time and by location within the study area. This module incorporates a chemical database with the physical and chemical properties of 469 substances, used in the species-by-species mortality calculations.

The biological effects module calculates losses to biological populations through time. The calculations include the following: the direct mortality to adult, juvenile, and larval biota due to toxic concentrations; recruitment losses due to stock effects; and the indirect mortality and weight loss to adult, juvenile, and larval biota due to the loss of foodstuff in the food web.

The economic damages module calculates the dollar values for injuries to biota based on use values. It also calculates the losses due to closures of fishing, waterfowl hunting, or beach areas.

The calculations rely on geographic data bases that contain average resource distributions for multiple habitat types within ten geographic regions throughout the coastal US, based on the classification scheme developed in Cowardin et al. (1979). Marine and estuarine systems are subdivided into subtidal and intertidal subsystems then broken down into additional habitat classes (based on shoreline type or bottom type). After the authors factored in the likelihood of each province-system-subsystem-class combination and the feasibility of collecting data for each likely grouping, they created a database with 36 intertidal and 55 subtidal ecosystem types with seasonal variations. Figure 2 provides a map of the ten regions, and Table 1 lists the habitat classifications.

The species in the database are classified into 13 categories, including nine fish categories. The nine fish categories represent 141 species, including both finfish and

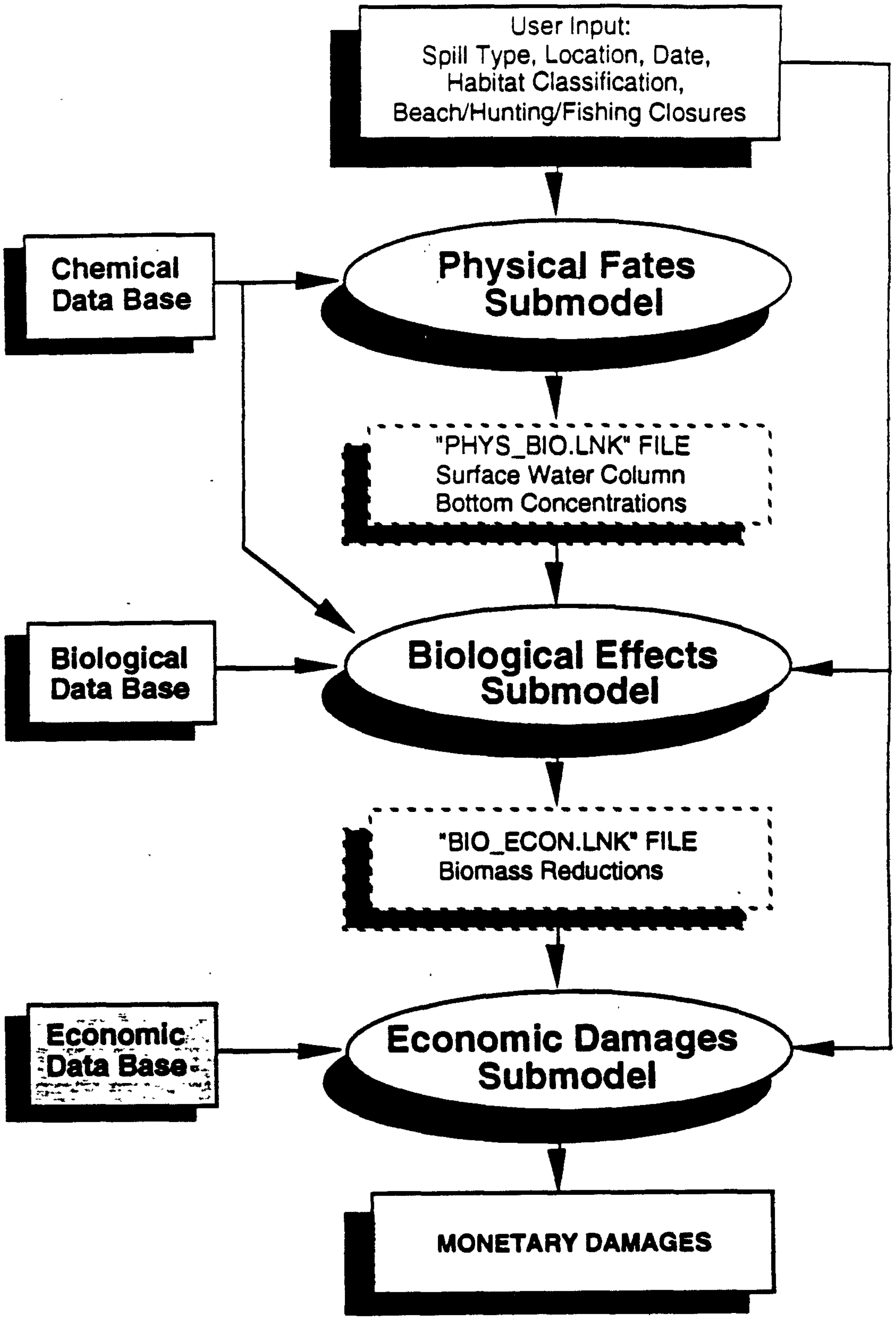

Figure 1. Model System Overview (NRDAM/CME)

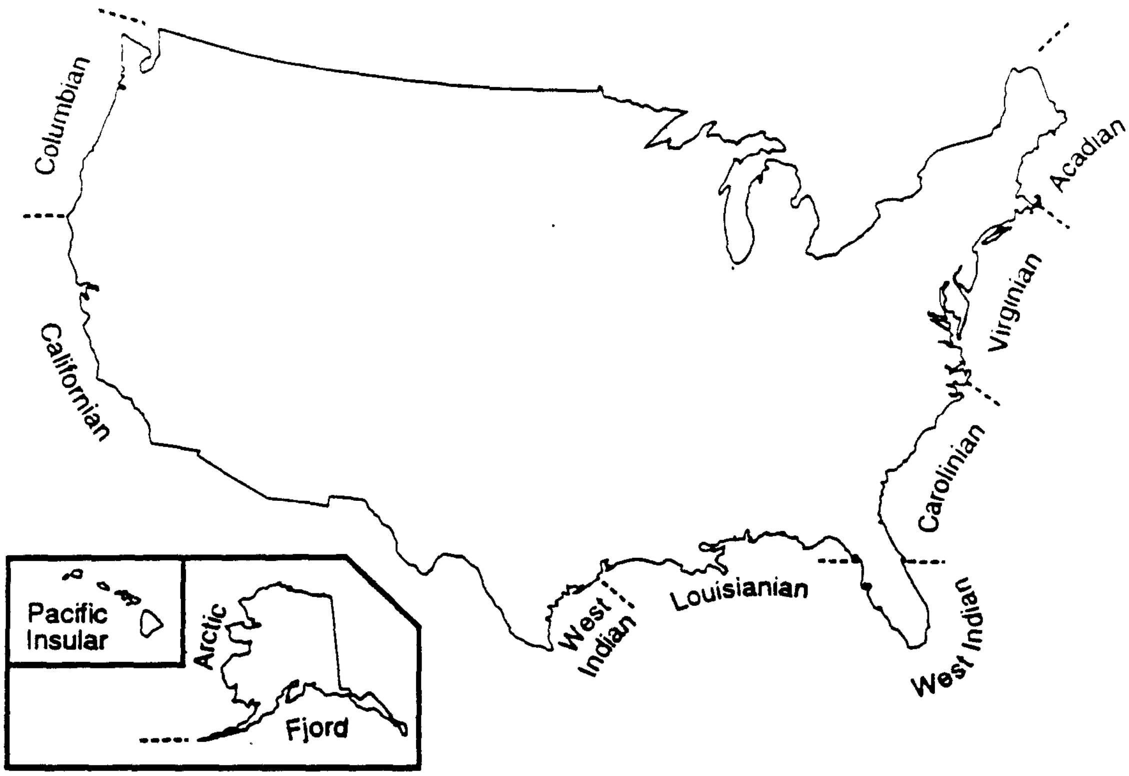

Figure 2. Boundaries of 10 Marine and Estuarine Provinces
Source: Type A Documentation

TABLE 1. HABITAT CLASSIFICATIONS

I. Ecosystem Types

A. 10 Marine and Estuarine Provinces

1. Atlantic and Gulf

P1. Acadian (Northeast: north of Cape Cod)

P2. Virginian (Mid-Atlantic: Cape Cod to Cape Hatteras)

P3. Carolinian (South-Atlantic: Cape Hatteras to Cape Canaveral)

P4. Louisianian (Gulf Coast: Cedar Key, Florida to Port Aransas, Texas)

P5. West Indian (South Florida, South Texas, West Indian Islands)

2. Pacific

P6. Californian (California: south of Cape Mendocino)

P7. Columbian (Pacific Northwest: Cape Mendocino to Vancouver Island)

P8. Fjord (Gulf of Alaska: south of Aleutian chain)

P9. Arctic (Alaska: North of Aleutian Chain)

P10. Pacific Insular (Hawaii and other Pacific islands)

a. Subtidal Bottom Types

S-B1. Rock bottom
S-B2. Cobble (unconsolidated)
S-B3. Sand (unconsolidated)
S-B4. Mud (unconsolidated)
S-B5. Rooted vascular aquatic bed (grasses)
S-B6. Macroalgal aquatic bed (e.g., kelp)
S-B7. Coral reef
S-B8. Mollusk reef
S-B9 Worm reef

b. Intertidal Bottom Types

I-B1. Rocky shore
I-B2. Cobbled beach
I-B3. Sandy beach
I-B4. Muddy shore
I-B5. Saltmarsh (cordgrass)
I-B6. Trees (coastal wetlands)
I-B7. Coral reef
I-B8. Mollusk reef

invertebrates (see Table 2). Four categories of species information are included: adult biomass, by species; larval numbers, by species category; mortality and growth parameters by species category; and primary and secondary productivity values.

The model is *not* intended to represent any specific localized populations of estuarine or marine situations: the databases represent *average* values for *representative* types of ecosystems. Consequently, to capture the necessary breadth of geographic coverage, the Type A Model has sacrificed geographic specificity.

CASE STUDY PROBLEM: VALUING RECREATIONAL FISH-KILLS AND FISHERY CLOSURES

Injury Quantification

Short-term (acute toxicity) losses are calculated separately for adults and larvae based on the toxicity information in the chemical database and the species distribution data. The model also calculates long-term losses due to the acute mortality to adult, juvenile, and larval biota due to toxic concentrations; the reduced recruitment into the adult fishery due to acute toxicity kills of larvae, juveniles, and adults; and the indirect mortality to adult, juvenile, and larval biota due to loss of foodstuff in the food web.

The fishery population dynamics in the model are based on the assumptions that the instantaneous catch rate (or catchability coefficient), the instantaneous natural mortality, and the growth function for individuals remain constant, and that egg production and larval numbers return to pre-spill levels immediately following dissipation of the spill. The architects of the model justify these assumptions on the grounds that the model is designed for spills of short duration.

Lost catch due to closure of an area to fishing is also calculated based on the biomass in the closed area. Because some of the lost catch in the closure area is due to mortality from acute toxicity, only the lost catch due to the closure in excess of the acute toxicity losses is added to the long-term losses to calculate total catch loss.

TABLE 2. SPECIES LIST AND CATEGORIZATION FOR BIOLOGICAL DATA SET

Species Number	Category[a]	Common Name	Scientific Name
1	1	American Shad	*Alosa sapidissima*
2	1	Alewife (and Blueback Herring)	*Alosa pseudoharengus, A. aestivalis*
3	2	Menhaden Atlantic and Gulf	*Brevoortia tyrannus, B. parronus*
4	2	Atlantic Herring	*Clupea harengus harengus*
5	2	Butterfish	*Peprilus triacanthus*
6	2	Pollock	*Pollachius virens*
7	2	Atlantic Mackerel	*Scomber scombrus*
8	3	Bluefish	*Pomatomus saltatrix*
9	3	Striped Bass	*Morone saxatilis*
10	3	Monkfish (Goosefish)	*Lophius americanus*
11	3	Weakfish (Grey Sea Trout)	*Cynoscion regalis*
12	4	Tuna	*Thunnus spp.*
13	4	Swordfish	*Xiphias gladius*
14	4	Sharks	Odontaspididae, Carcharhinidae, etc.
15	4	Dogfish	*Squalus acanthias*
16	5	Yellowtail Flounder	*Limanda ferruginea*
17	5	Summer Flounder (Fluke)	*Paralichthys dentatus*
18	5	American Plaice	*Hippoglossoides platessoides*
19	5	Witch Flounder	*Glyptocephalus cynoglossus*
20	5	Winter Flounder (Blackback)	*Pseudopleuronectes americanus*
21	6	Atlantic Cod	*Gladus morhua*
22	6	Haddock	*Melanogrammus aeglefinus*
23	6	Redfish (Ocean Perch)	*Sebastes fasciatus*
24	6	Silver Hake (Whiting)	*Merluccius bilinearis*
25	6	Red Hake	*Urophycis chuss*
26	6	White Hake	*Urophycis tenuis*
27	6	Scup	*Stenotomus chrysops*
28	6	Tilefish	*Lopholatilus chamaeleonticeps, Caulolatilus microps*
29	6	Black Sea Bass	*Centropristis striata*
30	6	Atlantic Wolffish	*Anarchichas lupus*
31	1	Hickory Shad	*Alosa mediocris*
32	2	King Mackerel	*Scomberomorus cavalla*
33	2	Spanish Mackerel	*Scomberomorus maculatus*
34	6	Harvestfish	*Peprilus alepidotus*
35	6	Atlantic Croaker	*Micropogonias undulantus*
36	6	Drums	*Sciaenidae*
37	6	Spot	*Leiostomus xanthurus*
38	6	Yellow Perch	*Perca flavescens*
39	6	Carp	*Cyprinus carpio*
40	6	Eels	*Anguilliformes*
42	2	Atlantic Thread Herring	*Opisthonema oglinum*
43	2	Anchovy, Atlantic	*Anchoa spp.*

(continued)

[a]Category Key

1 Anadromous fish
2 Planktivorous fish
3 Piscivorous fish
4 Top carnivorus
5 Demersal fish
6 Semi-demersal fish
7 Mollusks
8 Decapods
9 Squid
10 Mammals
11 Waterfowl
12 Shorebirds
13 Seabirds

TABLE 2. SPECIES LIST AND CATEGORIZATION FOR BIOLOGICAL DATA SET (CONTINUED)

Species Number	Category[a]	Common Name	Scientific Name
44	2	Striped Mullet	*Mugil cephalus*
45	6	Sheepshead	*Archosargus probatocephalus*
46	6	Spotted Sea Trout	*Cynoscion nebulosus*
47	6	Sand Sea Trout (White Sea Trout)	*Cynoscion arenarius*
48	6	Sea Catfish	*Arius felis*
49	3	Atlantic Halibut	*Hippoglossus hippoglossus*
50	3	Bonito (Tunny)	*Euthynnus alletteratus*
51	3	Crevalle Jack	*Caranx hippos*
52	3	Greater Amberjack	*Seriola dumerili*
53	3	Jacks, Other	*Carangidae*
54	3	Blue Runner	*Caranx crysos*
55	3	Dolphins	*Coryphaenidae*
56	5	Flounder, Southern	*Paralichthys lethostigma*
57	5	Flounder, Gulf	*Paralichthys albiqutta*
58	6	Drum, Red	*Sciaenops ocellatus*
59	6	Drum, Black	*Pogonias cromis*
60	6	Porgies	*Sparidae*
61	6	Florida Pompano	*Trachinotus carolinus*
62	6	Grunts	*Haemulidae*
63	6	Pinfish	*Lagodon rhombodies*
64	6	Kingfish	*Menticirrhus spp.*
65	6	Sheepshead	*Archosargus probatocephalus*
66	6	Cuck	*Brosme brosme*
67	6	Tautog	*Tutoga onitis*
68	6	Groupers	*Epinephelus spp., Mycteroperca spp.*
69	6	Snapper, Red	*Lutjanus campechanus*
70	6	Snapper, Other	*Lutjanidae*
71	6	Whiting (Southern Hakes)	*Urophycis floridanus*
72	2	Spanish Sardine	*Sardinella aurita*
73	6	Silver Jenny	*Eucinostomus gula*
74	6	Bonefish	*Albula vulpes*
75	3	Barracuda	*Sphyraenidae*
76	6	Sea Bass	*Serranidae*
77	6	Triggerfish	*Balistidae*
78	1	Salmon, Sockeye (= Red)	*Oncorhynchus nerka*
79	1	Salmon, Chum (= Keta)	*Oncorhynchus keta*
80	1	Salmon, Pink	*Oncorhynchus gorbuscha*
81	1	Salmon, Chinook (= King)	*Oncorhynchus tshawytscha*
82	1	Salmon, Coho (= Silver)	*Oncorhynchus kisutch*
83	2	Mackerel, Pacific	*Scomber japonicus*
84	2	Mackerel, Jack	*Trachurus symmetricus*
85	2	Anchovy, Pacific	*Engraulis mordax*
86	2	Herring, Sea (Pacific)	*Clupea harengus pallasi*

(continued)

[a]Category Key

1 Anadromous fish	5 Demersal fish	8 Decapods	11 Waterfowl
2 Planktivorous fish	6 Semi-demersal fish	9 Squid	12 Shorebirds
3 Piscivorous fish	7 Mollusks	10 Mammals	13 Seabirds
4 Top carnivorus			

TABLE 2. SPECIES LIST AND CATEGORIZATION FOR BIOLOGICAL DATA SET (CONTINUED)

Species Number	Category[a]	Common Name	Scientific Name
87	5	Flounder, Pacific	*Pleuronectidae*
88	5	Halibut, Pacific	*Hippoglossus stenolepis*
89	6	Perch, Pacific Ocean	*Sebastes alutus*
90	6	Rockfish, Other	*Sebastes spp.*
91	6	Perch, Other	*Embiotoca spp., Amphistichus spp., Hyperprosopon spp.*
92	6	Sablefish (Black Cod)	*Anoplopoma fimbria*
93	6	Cod, True (Pacific)	*Gaus macrocephalus*
94	6	Lingcod	*Ophrodon elongatus*
95	6	Hake, Pacific (Whiting)	*Merluccius productus*
96	6	Sea Bass	*Serranidae*
97	2	Pollock, Walleye	*Theragra chalcogramma*
98	2	Mackerel, Atka	*Pleurogrammus monopterygius*
99	5	Sole, Yellowfin	*Limanda aspera*
100	5	Flounder, Arrowtooth	*Atheresthes stomias*
101	5	Turbot, Greenland	*Reinhardtius hippoglossoides*
102	5	Plaice, Alaska	*Pleuronectes quadrituberculatus*
103	7	Smelt	*Osmeridae*
104	6	Flounder, Starry	*Paralichthys stellatus*
105	6	Sole, Butter	*Isopsetta isolepis*
106	6	Sole, Dover	*Microstomus pacificus*
107	6	Sole, English	*Parophtys vetulus*
108	6	Sole, Rock	*Lepidopsetta bilineata*
199	6	Other Fish	*(generic)*
Invertebrates			
201	7	Surf Clam	*Spisula solidissima*
202	7	Ocean Quahog	*Artica islandica*
203	7	Atlanta Sea Scallop	*Placopecten magellanicus*
204	8	American Lobster	*Homarus americanus*
205	8	Northern Shrimp	*Pandalus borealis*
206	8	Red Crab	*Geryon quinquedens*
207	9	Squid, Atlantic	*Loligo pealei, Illes illecebrosus*
208	7	Blue Mussel	*Mytilus edulis*
209	8	Blue Crab (Hard Shell)	*Callinectes sapidus*
210	8	Blue Crab (Soft Shell)	*Callinectes sapidus*
211	7	Son Clam	*Mya arenaria*
212	7	Oyster, Atlantic	*Crassostrea virginica*
213	7	Hard Clam (Quanog)	*Mercenaria mercenaria*
214	7	Conch	*Strombus spp.*
215	8	Shrimp (Brown, Pink, White)	*Penaeus spp.*
216	7	Calico Scallop	*Argopecten gibbus*
217	8	Crabs (general)	*(generic)*
218	8	Stone Crab	*Menippe mercenaria*

(continued)

[a]Category Key

1 Anadromous fish	5 Demersal fish	8 Decapods	11 Waterfowl
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4 Top carnivorus			

TABLE 2. SPECIES LIST AND CATEGORIZATION FOR BIOLOGICAL DATA SET (CONTINUED)

Species Number	Category[a]	Common Name	Scientific Name
219	8	Lobster, Spiny	*Panuliris spp.*
220	7	Abalone	*Haliotis spp.*
221	8	Crab, Dungeness	*Cancer magister*
222	8	Shrimp, Pacific	*Pandalus borealis*
223	9	Squid, Pacific	*Loligo, opalescens, Berryteuthis magister, Onychoteuthis boreali japonicus*
224	8	Crab, Snow (Tanner)	*Chionoecetes*
225	8	Crab, King	*Paralithodes camtschatica, P. platypus*
226	7	Clam, Butter	*Saxidomus nuttalli*
227	7	Clam, Horse	*Tresus capax*
228	7	Clam, Geoduc	*Panopea generosa*
229	7	Clam, Manila	*Tapes philippinarum*
230	7	Oyster, Pacific	*Crassostrea gigas*
231	7	Oyster, Olympic	*Ostrea lurida*
232	7	Atlantic Bay Scallop	*Argopecten irradians*
233	7	Pacific Sea Scallop	*Pecten caurinus*
299	7	Other Invertibreates	(generic)
Birds			
301	11	Marsh Ducks	Anatinae
302	11	Diving Ducks	Aythyinae
303	11	Mergansers	Merginae
304	11	Whistling Ducks	Dendrocygninae
305	11	Stiff-Tailed Ducks	Oxyurinae
306	11	Coots	Rallidae
307	11	Geese	Anserinae
308	11	Swans	Cygninae
311	12	Sandpipers	Scolopacidae
312	12	Plovers	Charadriidae
313	12	Turnstones	Aphrizidae
314	12	Oyster Catchers	Haematopodidae
315	12	Phalaropes	Phalaropodidae
316	12	Avocetes, Stilts	Recurvirostridae
321	13	Gulls, Terns	Laridae
322	13	Cormorants	Phalacrocoracidae
323	13	Auks	Alcidae
324	13	Shearwaters	Procellariidae
325	13	Storm Petrels	Hydrobatidae
326	13	Pelicans	Pelecanidae
327	13	Frigatebirds	Fregatidae
328	13	Gannets, Boobies	Sulidae

[a]Category Key

1 Anadromous fish
2 Planktivorous fish
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Valuation of Damages

Translation from Change in Stock to Change in Trip Catch and Number of Affected Trips

In the biological submodel, the fish stock is allocated to recreational catch mortality, commercial catch mortality, and natural mortality based on share parameters for each species in the database. The predicted reduction in stock due to a spill is also allocated to those categories, assuming constant proportions. Jim Opaluch, one of the authors of the economic module (and a participant in the case study group), indicated that an assumption implicit in the valuation procedure was that all species are highly mobile; with this assumption, the change in fish stock will be spread over a wide geographical area and generally will produce a small change in catch rate (trip quality) over a large number of trips.

The value per fish, catchability coefficient, level of fishing effort, and cost per unit effort parameters are assumed to be unaffected by the spill. Consequently, the decline in recreational fishing catch due to a spill is calculated as the recreational fishing share of the stock (a parameter in the database) times the change in the fishery stock calculated in the biological module.

Valuation of the Change in Catch Rates

The valuation procedure then assigns the reduction in recreational stock size at a rate of one fewer fish per angler. In the calculation, the number of anglers affected just equals the change in the recreational stock size; there is no independent calculation of total trips affected. This procedure is a creative way to avoid explicitly characterizing the levels of fishing participation affected by the spill (which is likely to be larger than the spill area because of fish mobility).

To generate the recreational fishing values for the Type A model, the authors relied on two studies providing an estimate of the change in the value of recreational fishing trips with a unit change in catch rate. Rowe et al. (1985) provide consumer surplus estimates for trips to California, Oregon, and Washington marine fisheries from separate random utility models for each state. For selected species, the scenario valued was the increase in the catch rate of one species by one fish/trip at all site/mode combinations where the species is caught Norton, Smith, and Strand (1983) provide estimates of the changes in consumer surplus with changes in catch rates for several striped bass fisheries on the East Coast. They employed a single-equation travel cost model.

Because these two studies valued only a few species, the modelers needed a procedure to provide values for other species. They calculated the change in consumer surplus on a weight basis for the available species. Judging that the variation in the value per pound did not appear to vary greatly across the species valued in the studies, they employed the simple mean of the estimates ($1.84/lb) in the model to value losses of all species.

QUESTIONS DISCUSSED IN THE CASE STUDY SESSION

We discussed whether the current procedures for valuing recreational fishing injuries in the Type A model can be improved. We considered the adjustments that would contribute the most to improving the estimates and the adjustments that are currently feasible.

The group proposed separate discussions of the injury from fish kills, which we believed was appropriately valued as a change in quality of the recreational fishery, and the injury from fishery closures, which we thought might better be modeled as a change in the quantity of resources available. We consider each modeling context separately below. For most possible extensions, we concluded that data are insufficient to determine whether such changes would represent substantial refinements to the model calculations. The discussion produced a series of recommendations for further research. In the final section, we discuss criteria to be used in selecting studies for inclusion in the model database.

MODELING ISSUES

Population Effects Due To Fish Kills And Their Impact On Fish Population Dynamics

Currently, the effect of fish kills is modeled as a change in the quality of recreational fishing trips that affects the trip value but does not affect total participation in the fishery. A single value per gram of fish killed appears in the model: the variation across species in damages per fish killed is completely driven by variation in average weight across species. In addition, the value does not vary with the size of the spill (and the effect on stock and catch rates) or the extent to which available substitutes are similarly affected. We discussed several possible extensions to the modeling, as reported below.

- Expand the single recreational fish value included in the database to a matrix of values, including variations in the value of lost fish by

  -fish species,

  -geographical area of spill, and

  -user types.

Most members of the group thought incorporating species and geographical variations could be an important contribution to the model and believed that some additional values have appeared in the literature since the model was first developed. We did not think that incorporating variations in consumer surplus values by user types would make an important contribution.

- Adapt the modeling and expand the value database to incorporate variations in the change in consumer surplus per unit change in catch depending on

  -the level of the change in catch per trip (i.e., avoiding the assumption that the change in consumer surplus is linear in catch); and

  -the extent to which substitutes are affected (which will vary substantially depending on whether the affected species have localized populations or are highly mobile over a wide area).

To implement either, it would be necessary to change the modeling to identify the geographic zone of impact (taking into account the mobility of the species) and the number of trips taken to that zone. With this information, an estimated change in catch per affected trip could be calculated (rather than implicitly assigning a reduction of one fish per trip.). In addition, the Type A model would need a matrix of values in the database, capturing the nonlinearities and substitution possibilities in the values.

Are the size of the change in catch per trip and the extent of the substitutes affected important sources of variation in value? The group discussion was inconclusive: we concluded that research is needed to explore these issues. To the extent that spills valued with the model are relatively small and the species are mobile, nonlinearities in the change in consumer surplus with a change in catch rates are not likely to have a large effect on values. For spills heavily injuring highly localized species, the variation in the change in catch rate may be much greater; for this context, exploring the possibility of substantial nonlinearities is more important. Impacts on localized groupings of species also raise questions regarding the treatment of variations in substitution possibilities.

Some preliminary analysis by Graham-Tomasi and Sung with the Michigan recreational fishing model (Jones and Sung, 1991) suggests that variation in substitution possibilities has a far greater effect on the value per lost fish than variation in the quantity of fish lost per trip.

Are these changes feasible? Unfortunately, we had serious questions about the availability of necessary data. The NMFS marine recreational surveys were cited as a possible source of data on trips. In addition, we discussed how to implement the variations in value with nonlinearities and substitution possibilities. Because of the difficulty of establishing a formula, some individuals in the group suggested creating categories of “small/medium/large effects” and assigning spills to suitable categories. However the distinctions are to be implemented, additional research needs to be done to generate the necessary values for making such distinctions.

- Incorporate changes in fishing participation as a result of spill-induced quality changes in the fisheries.

Currently, the Type A model treats fishing participation levels as constant when fishing quality changes based on the assumption of mobile fish species. With this assumption, the population changes generally being modeled would yield small changes over a wide geographic area. We concluded that further research would be useful to identify how elastic trip participation is to quality changes (at the level of quality changes involved) and the extent to which damages are underestimated by excluding this category of effects.

Some recent preliminary analysis of the Michigan recreational fishery model performed by Graham-Tomasi and Sung indicates that, though the participation elasticity is not large, the share of damages contributed by that behavioral response may be substantial.

Incorporating this extension in the model would require developing a generic participation equation. Before this equation could be added, we would need to include the modeling and database adjustments required to implement it. Those adjustments would build into the model the capacity to identify the zone of impact on the fisheries (taking into account fish mobility) then determining the impact on trip catch in the affected zone and the total number of trips in the affected zone.

An additional requirement would be to ensure that the modeling in the fishery dynamics and the valuation portions of the model are consistent regarding trip participation. We believed ensuring this consistency would not be difficult.

Fishery Closures

Fish not caught because of a closure are valued using the same procedures as for fish kills, that is, the total number of trips is assumed constant, but the value of each affected trip is reduced because of the lower catch rate. This procedure implicitly assumes a small closure area and the existence of (perfect) substitute sites sufficiently nearby so that additional travel costs are essentially zero.

We believed considering modeling closures as a change in quantity of fishing resources would be appropriate. In this case, the correct calculation of damages for a change in quantity of recreational fishing services would be the change in trips times the consumer surplus per trip. Ideally, in the studies providing the basis for the consumer surplus of a lost fishing trip, the species and site characteristics are similar to the closure area, and the substitution possibilities are similar in both study and spill contexts.

This extension would seem to be more important in cases in which most close substitution opportunities are not available. The current procedures appear adequate in cases of a small area of closure.

Incorporating this extension would require trip participation rates and additional consumer surplus values on a per-trip basis. More studies are likely to be available for valuing fishing trips (as needed in this extension, modeling a change in quantity) than for valuing changes in the catch rate on trips (as needed for a change in quality).

RECOMMENDATIONS FOR FURTHER RESEARCH

We generally felt that additional work is needed to explore whether substantial variations exist in consumer surplus for a change in catch per trip by species, geographic area, size of the effect, and the extent of substitution possibilities that are affected. The group agreed that the current set of random utility models that have been estimated provides a good basis for such analysis. The participation question also needs to be explored; this research can be done with the participation models linked to random utility models or with the earlier generation travel cost models, employing equations estimating total trips.

SELECTION OF STUDIES FOR INCLUSION IN THE DATABASE

The selection of studies and specific consumer surplus value calculations from the studies is critical to the model database. We addressed the following issue: What criteria should be applied to exercise quality control in the choice of studies used to estimate consumer values? We identified three sets of criteria that may be relevant to the selection of studies:

- relevance of the consumer surplus measure to the context (change in quality, loss of access)
- quality of study (meets minimum standards)
- comparability of context between the study site and the spill site

However, we did not agree on how to apply the criteria. We did not believe that the current literature provides enough basis to decide what factors are operationally important in determining "comparability." And we concluded that the quality judgment needs to be made within the context of the study's objective and its use in the transfer.

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LONG-TERM HEALTH RISKS VALUATION: PIGEON RIVER, NORTH CAROLINA

Susan B. Kask*

ABSTRACT

Executive Order 12291 requires benefit-cost analysis for all government legislation. Does this mean that for each piece of environmental legislation we must provide new health benefits estimates for each illness and each toxin to value benefits? Estimating the benefits of a reduction in health risks is a difficult task for the policy researcher. In this paper we present a protocol for transferring health benefits from a study site to a different policy site and provide an example of its application.

Protection of public health is a primary goal of much of U.S. environmental legislation because environmental pollution can have a variety of negative effects on public health. For example air pollution can cause itchy eyes, chronic respiratory disease, and even death for those most sensitive. These effects, however, occur with some probability. Environmental pollution increases the risk of exposure to a contaminant, which in turn increases the risk of adverse health effects (see Figure 1). A benefit from reduced pollution is the reduction in the risk of these health effects. To evaluate the benefits from environmental pollution control legislation, we must account for these health benefits.

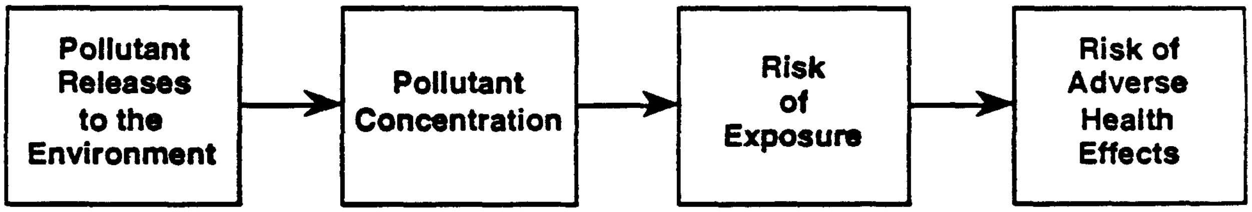

Figure 1. The Link Between Pollution and Health

Estimating the benefits of risk reduction is difficult for the policy researcher. How much individuals value a reduction in their future risk of contracting cancer or chronic illness from a reduction in pollution is a challenge to estimate. Furthermore, estimating the value of reduced

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risk of acute illness or discomfort from a variety of symptoms is equally problematic. Must we provide a new estimate for each illness, for each toxin, to value benefits? Studies exist that value accidental death, death at some future date, and reductions in illness days, for example. Can we use these studies as proxy estimates across illnesses and toxins? Can they be transferred spatially? This paper explores the potential to transfer health benefits.

We present a basic model underlying health benefit estimates. We also present the primary issues and a proposed protocol for benefits transfer. To demonstrate the protocol and illustrate the pitfalls of transfer, we consider a case study. Finally we present our conclusions and recommendations for future research.

CONVENTIONAL THEORY OF HEALTH BENEFITS MEASUREMENT

The typical model for measuring health benefits usually begins with a damage or production function that links self-insurance activities (e.g., medical treatment, purchase of air conditioners, diet, and exercise) to health. We denote this function as

$$H = H(Z)$$

where Z is a vector of self-insurance activities and H is a state of health. In some cases H is also a function of the level of pollutant (Shogren and Crocker, 1991). The production function may be represented with a two-state model with state 0 representing good health and state 1 representing death (Smith and Desvousges, 1987), or alternatively, H may represent an index or a continuum of health outcomes (Dickie and Gerking, 1991; Shogren and Crocker, 1991). Here we assume a two-state world for illustrative purposes.

As shown in Figure 1, pollution affects health through the risk of exposure and the risk of adverse health effects given exposure. We can include pollution into a probability density function Q, representing the probability of having good health. This probability depends on the level of pollution in the environment, which in turn affects the level of exposure of an individual, and the individual's level of private self-protection. This probability function is

$$Q = Q(X, Q)$$

where X is the level of private self-protection and Q is the level of some pollutant in the \environment. An alternative approach is found in Smith and Desvousges (1987) where they separate the risk of exposure and the risk of illness, and the level of pollutant affects the risk of exposure.

Each individual has an indirect utility function

$$V = V[M, H(Z)]$$

where M is their income and H is their level of health. In a two-state world where H_0 is good health and H_1 is poor health, consumers maximize expected utility given some level of pollution

$$\pi (X, Q_0)V[M, H_0(Z)] + [1\text{-}Q(\pi, Q_0)]V[M, H_1(Z)].$$

Their willingness to pay (WTP) for a small change in Q given self-protection is the difference between the level of utility in each state divided by the expected marginal utility of income.

$$WTP = \frac{V(M, H_0) - (M, H_1)}{\pi V_M^0 + (1-\pi) V_M^1} \cdot \frac{\partial \pi}{\partial Q} dQ$$

Alternatively, a discrete decrease in Q from Q_0 to Q_1 is represented as

$$\pi(X, Q_0)V(M, H_0) + [1 - \pi(X, Q_0)]V(M, H_1) =$$

$$\pi(X, Q_1)V(M-P, H_0) + [1 - \pi(X, Q_1)]V(M-P, H_1)$$

where P represents the WTP[1] to maintain the initial level of utility at the new level of pollution (a Hicksian compensating measure of welfare change). Using a variety of benefits estimation techniques, we can estimate the value of P given self-protection expenditures.

A PROTOCOL FOR HEALTH BENEFITS TRANSFER

The overriding concern for public health behind much of U. S. environmental legislation, and Executive Order 12291 suggests a significant demand exists, and will continue to exist, for benefit estimates of reduced risk to health. Evaluation of these benefits will require expensive and time-consuming projects for each substance and health effect. Benefits transfer may provide a solution to satisfying the need for benefits analysis for the variety of environmental legislation and regulation in the U.S. However, the transfer approach poses potential risks: poor quality

[1] Smith and Devousges (1987) refer to this value as an option price.

benefits transfers may lead to incorrect policy choices (Desvousges, Naughton, and Parsons, 1992). A sound approach to transfer is necessary.

Benefits transfers apply existing benefit estimates from a study site to a policy site. Researchers must transfer the issue or commodity from a particular policy site into something that can be interpreted using existing information (Smith, 1992). What criteria should we use to transfer health benefits from a study site to a policy site? Table 1 lists our general recommended approach for a transfer analysis. We focus on Stage 2, Transfer Criteria, in more detail below. We identify three areas as the primary focus for a transfer protocol: commodity specification, market and exchange mechanism, and site and sample characteristics. We discuss each below.

TABLE 1. GENERAL APPROACH FOR TRANSFER ANALYSIS

- Define the purpose of the estimates and the level of precision needed.
- Use proposed transfer criteria (commodity, sample, market, site) to describe study site.
- Select an existing benefit study or studies that satisfy the transfer criteria, keeping in mind estimates' purpose and precision.
- Determine the appropriate transfer method (e.g., point estimate or confidence interval, function transfer, Bayesian approach, or meta-analysis).

The Transfer Protocol: Commodity Specification

One of the most important steps in a benefits estimation and benefits transfer is careful specification of the commodity to be valued. How should we define our commodity when valuing health benefits? Table 2 identifies six areas for clarification in commodity specification.

Response/Causal Agent: Should we define our commodity based on the substance or the end result (morbidity/mortality or both)? We recommend that the commodity in health transfer studies be defined by the end result, the risk of illness or death. We posit that ultimately the consumer cares about the health effect (i.e., the itchy eyes, coughing, birth defects) and not so much the source or pollutant that causes the health effect. If this position proves defensible, then benefits transfer exercises become significantly less complicated because we can consider reductions in cancer risk from exposure to benzene in the air, for example, the same as a reduction of cancer risk from dioxin exposure in the water. This position, however, may not hold true for pollution sources that have variations in avoidance opportunities and, as discussed in

TABLE 2. RECOMMENDED COMMODITY SPECIFICATION CRITERIA

Response/Causal agent	Should we define our commodity based on the substance or the health effect?
Risk definition	Are we changing risk through changes in probability, in severity of a health effect, or both?
Temporal dimensions	Is there a latency period between exposure and occurrence of health effect?
Voluntary and involuntary dimension	Is exposure voluntary or involuntary?
Exposure pathway	Does exposure occur through water, air, and food, for example?
Exposure level	Is exposure cumulative or acute?

more detail below, morbidity effects. Thus, the role of the causal agent in risk valuation responses is an important research issue.

If we base our commodity specification on the end result, the illness, we then should consider the potential to transfer values across illnesses. For example, can we transfer the health benefit estimates for a reduction in the risk of death from lung cancer to liver cancer? To best answer this question let us consider the three general categories for valuation in health benefit studies: death, illness with no death, and illness followed by death. In the first case, individuals value mortality alone. A pure morbidity value is provided in the second case and a combined value in the third. Returning to our question above, an individual may not value death from lung cancer the same as death from liver cancer, because this is actually a combined value and the morbidity characteristics may vary across disease. Variation in morbidity across diseases may include differences in severity or timing for example.

This potential for variation in morbidity characteristics may also cause problems for transfer across pollutant sources for the same disease. For example, consumers may value reduced risk of lung cancer from dioxin exposure the same as reduced risk of lung cancer from asbestos, only if the morbidity characteristics and avoidance opportunities are the same between causal agents.

Symptoms and the potential for death should be the primary factors used to define the commodity in a health benefits transfer study. However, the pollutant source may be more important if avoidance opportunities, or morbidity effects, vary across sources. The cause of the symptoms, or death (e.g., lung cancer versus liver cancer) may also be important to value estimates because morbidity characteristics may vary.

Although we have three general categories for valuing health benefits, no studies have yet valued combined mortality and morbidity impacts. We recommend researchers use mortality estimates as lower bounds in the absence of combined studies. Because morbidity is already an element in these measures, adding morbidity and mortality values may result in double counting. Finally, the units of measurement for the commodity defined are important. If health risks are portrayed as unit days of a symptom, the researcher must consider the problems of over or under estimation surrounding unit day measures (Morey, 1992).

Risk Definition: Environmentally related health effects can range from acute illness and discomfort, which may occur with a high probability, to sudden death that may occur with a low probability. The components of risk include both the probability of a health effect occurring as well as the severity of that health effect. Ehrlich and Becker (1972) recognize that risk can be reduced by decreasing either element. In a laboratory environment, Shogren (1990) found reductions in probability were preferred to severity reduction. Whether policy changes the severity of the event or the probability of its occurrence can influence how consumers value a change in the overall risk. Therefore, when evaluating study and policy sites, researchers must clarify the component of risk that the proposed policy is changing-probability or severity. Secondly, considering the direction and magnitude of the risk change is important. Does the probability or severity of the policy under consideration increase or decrease? In the absence of information on symmetry, researchers should be cautious in transferring the health benefit estimates from an increase in probability at a study site to a policy site where a decrease in probability occurs.

Temporal Dimensions: Health effects from environmental hazards range from acute immediate effects to chronic latent health effects. The temporal dimension of health effects includes the length of time the illness occurs and the time period between exposure and occurrence of the illness or death. We cannot assume that consumers will value latent health effects the same as immediate effects nor assume they would value chronic and acute effects in the same fashion. Therefore, looking for similarities in the temporal dimensions of the health effects between the policy site and the study site is important. Presumably, temporal dimensions are similar when the health effect is constant across sites.

Voluntary and Involuntary Dimension: Although we have stated that the pollutant or source of a disease may be unimportant when transferring health benefit estimates, in one case characteristics of the source become important: the voluntary/involuntary nature of exposure to a health hazard. Environmental health risks are typically involuntary (a person is unknowingly exposed) as compared to health risks from smoking, drinking, and driving, for example (a person chooses to incur the risk). Valuation of voluntary risks may be quite different from involuntary (Starr, 1969; Starr, 1979); thus they should not be used interchangeably. The distinction occurs because voluntary risks imply some form of control over the risk, and perceived control can influence the value of risk reduction.

Exposure Pathway: Although we have ruled out the importance of the pollutant's source in value estimates, we may find that the exposure pathway affects consumer values. This effect would become relevant if exposure pathways influence our ability to avoid a hazard or the voluntary nature of exposure. For example, individuals may perceive greater control over the quality of their water and food than over air quality.

Exposure Level: Exposure to environmental pollutants can range from short time periods with high doses to long time periods with low doses. How consumers value a change in health risk will be influenced by these exposure levels, because they influence consumer probability perceptions and time preferences. Therefore, researchers must choose study sites with similar exposure levels as policy sites for benefits transfer.

Transfer Protocol: Sample and Site Characteristics

Researchers classify sample and site characteristics in two general areas: the socioeconomic characteristics of the sample and the location and temporal characteristics of the site. Characteristics that should be highlighted in a health benefits transfer study are discussed below.

Socioeconomic Characteristics: Sample characteristics such as income, education, age, awareness of risk, baseline health, and baseline risk may affect benefit estimates. Because the sample in a study site is probably not identical to the policy site, researchers must find study site value estimates that have well-developed valuation models. These models should include the socioeconomic factors that influence estimates and thus provide more insight into the relationship between demographic characteristics of the sample and values estimated. Good understanding and documentation of study site demographics will allow researchers to identify the sample characteristics that vary across study and policy sites.

Location and Temporal Characteristics: Just as socioeconomic characteristics affect benefit estimates, the researcher must also be aware of certain site characteristics that influence values. For example, location characteristics possibly important to health benefits estimation include the presence of insurance programs, access to medical care, potential for avoidance opportunities, climate, time period of exposure, and baseline exposure levels. The analyst should establish a relationship between these location and temporal characteristics and the values given at the study site. As above, reporting of these characteristics for the study site is important. Finally, as with an original benefits estimation study, analysts must consider the size of the population affected to calculate total benefits.

Transfer Protocol: Market and Exchange Mechanisms

Psychologists discovered that alternative means of framing a problem can systematically influence choice and values (e.g., Tversky and Kahneman, 1981). Three important factors regarding framing effects of a risk valuation problem are the risk reduction technology, the exchange medium, and the type of question (WTP/willingness to accept [WTA]). Finally, an additional market issue is the presence of nonuse values in the market. The importance of these issues for benefits transfer is discussed below.

Risk Reduction Technology: Evidence suggests that alternative risk reduction strategies influence valuation. Individuals can produce a given reduction privately or collectively. Individual preference for private or collective reduction depends on the payment's perceived productivity. Collective reduction may prove more efficient given scale economies, because many private actions are too expensive or complicated to be economically feasible (Shogren, 1990). However if excessive free-riding is perceived, private reduction may be valued more highly. Thus, determining the risk reduction strategies most appropriate for the policy site is important. Figure 2 illustrates the individual's choice of risk reduction actions.

Exchange Medium: One of the most important factors in designing a valuation study is the exchange medium (or "payment vehicle"). Consumers can pay to reduce the risk of adverse health effects through wages, taxes, or prices. The medium can influence values given; thus using a realistic medium for the policy site is important for both benefits transfer, as well as original benefits studies.

Nonuse Values and WTP/WTA: Analysts must determine whether nonuse values are relevant and what welfare change measure is appropriate for the policy site. Nonuse values include the health effects of children, other relatives, neighbors, and friends. Consumers may value the health of others as well as their own health. However, the extent to which these nonuse

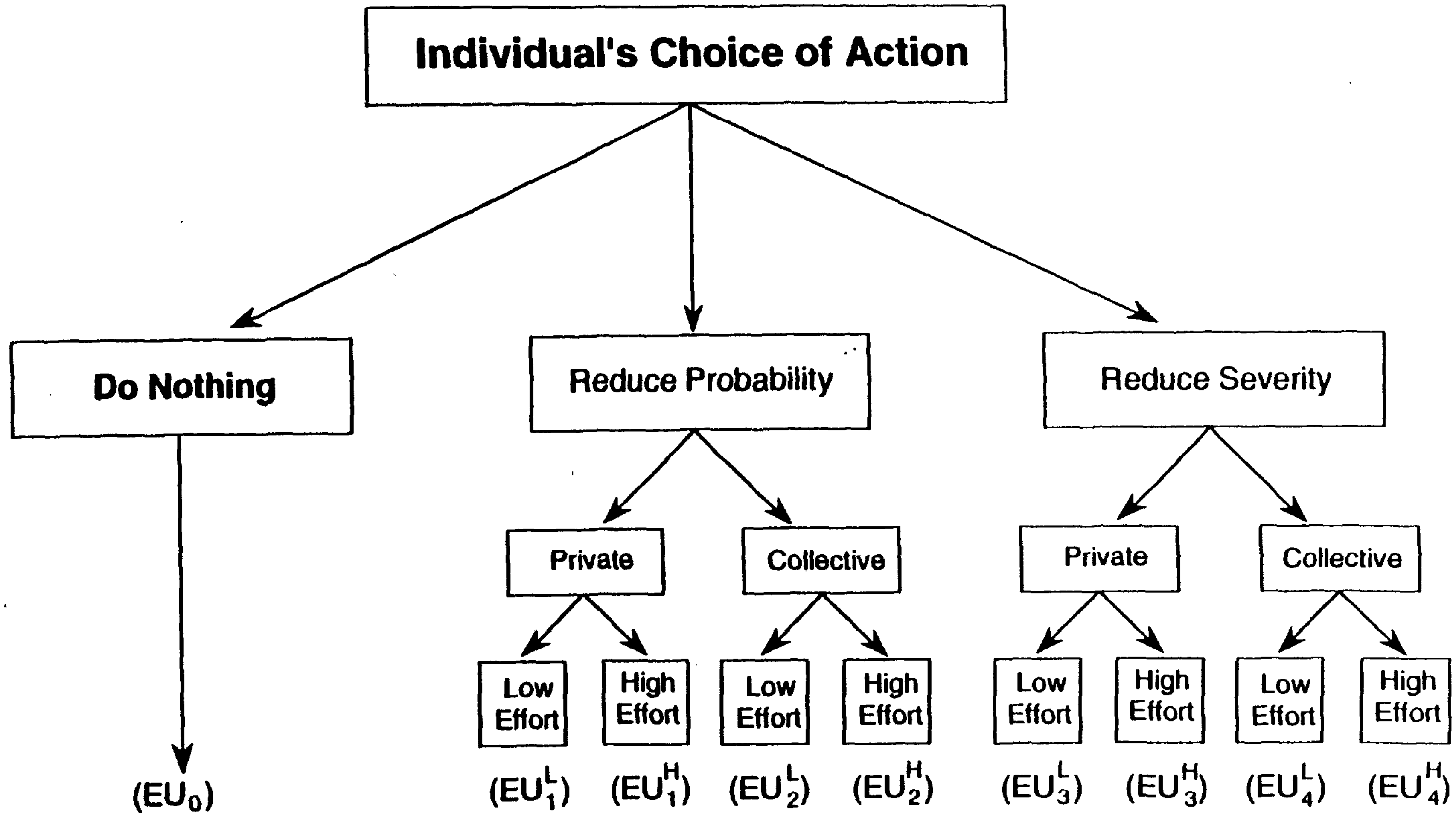

Figure 2. Individual's Choice of Risk Reduction Actions

values may be embedded within current value statements given by individuals is unclear. Although not readily available, some measure of nonuse values might be appropriate in health transfer studies.

Selecting between Hicksian compensating and equivalent measures and using WTP or WTA depends on the property rights allocation and the direction of the policy change for the particular policy site. Therefore, well-defined property rights and risk reduction should be consistent across the sites. Otherwise, extrapolating one value measure for another is questionable given the theoretically predicted and empirically observed divergence in WTP and WTA for improved health quality.

Study Selection

Following the transfer protocol suggested above, an analyst can select the study sites most appropriate for valuation at the new policy site. We recommend that existing contingent valuation method (CVM) studies be given priority because the alternative approaches have an array of problems. CVM studies are preferred because of their potential to capture morbidity and the diversity of possible samples (i.e., general population versus white male workers).

If CVM studies are unavailable, we recommend the few averting behavior studies and experimental laboratory studies. Hedonic wage models are given a lower priority because of the narrow sample group and the focus on risk of accidental death. Cost of illness is given the lowest priority because of its weak theoretical underpinning.

Additional selection criteria may include the theoretical soundness of the study, level of information reported, and purpose of estimates and level of precision required. Of course the study site should match policy site specifications to a level the researcher considers acceptable.

A CASE STUDY: LONG-TERM HEALTH RISKS FROM SURFACE WATER POLLUTION

A classic case of exposure to a long-term health risk is found in Western North Carolina. Champion Paper currently discharges approximately 43 million gallons of coffee-colored wastewater into the Pigeon River daily. In addition to the discoloration, a potentially more serious problem is the risk to public health from the dioxin and other toxins present in the discharge. The state of North Carolina is considering a weakening of the maximum allowable dioxin limit of 14 parts per trillion (ppt). What are the benefits of maintaining the limit or the costs of raising the limit? This case study provides a working example of the need to transfer benefit estimates and the many potential problems for the valuation of changes in long-term health risks from surface water contamination.

The Site: The Pigeon River originates in Haywood County, North Carolina, as a pristine stream in the Pisgah National Forest. The river flows north, 10 miles, to Canton, where Champion paper discharges their effluent. The river continues northwest, 16 miles, crossing the Tennessee state border past seven small communities in both states until it reaches Newport, in Cocke County, Tennessee. Thirty-six miles from the mill, the river empties into Douglas Lake. The 1990 mean flow rates, north of Canton, vary from a low fall flow of 88 cfs to a high of 10,900 cfs in the spring. The river is regulated by Lake Logan and Walters Lake.

The Pigeon flows through mountainous terrain between the Great Smokey and Bald Mountains. The river above Canton is used both as a municipal drinking water source, rated WS3, and for recreational activities such as swimming, boating, and fishing. Downstream from Canton, the river has been rated as Class C water for boating and fishing only; immersion is not recommended. A posted advisory recommends against eating fish caught in the river north of Canton. The 10-mile stretch from Walters Lake to the state line is considered a good "brown" water rafting run and is sometimes used by recreationists in the area. In Tennessee, the river is classified and protected for industrial water use, fish and aquatic life, recreational activities including swimming, irrigation, and livestock and wildlife watering. But, because of the present level of discharge the river does not meet state requirements for aquatic life or recreational uses. Tennessee has posted a warning against eating fish from the river. In addition, the present high color level prohibits any additional waste discharge; thus the river is not used for any other industrial discharge in Tennessee.

Water Contamination: In 1989, industrial water use accounted for 85.6 percent of water used in Haywood County. Fifty-one percent of industrial water is used by Champion Paper in a pulp mill[2], paper mill,[3] and their utilities and filter plants.[4] They produce food board and fine paper using an integrated bleached kraft pulp and paper manufacturing process.

Pollutants present in the discharge in either significant quantities or regulated by EPA are given in Table 3. In addition to the pollutants in Table 3, the discharge also affects the stream's temperature and acidity. The average winter effluent temperature is 29.8°C and the summer temperature is 37.9°C. Acidity levels range from pH 6.4 to 8.2. EPA temperature limits for effluent are between 29° to 32°C, with a 13°C maximum increase in stream temperature. The acidity limits are pH 6 to 9.

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[2]Includes chip cooking, pulp washing, screening and bleaching, recovery and generation of cooking chemicals, and production of chlorine dioxide for bleaching.
[3]Produces fine paper, food board, and dried pulp.
[4]EPA Forms 1 and 2C submitted by J. R. Kilpatrick to EPA Region IV Office, Atlanta, GA.

TABLE 3. DISCHARGE POLLUTANTS FOR CHAMPION PAPER MILL IN CANTON, NORTH CAROLINA (1989)

	1989 Sample Values		
Effluent Characteristic	**Daily Average**	**Daily Max**	**Daily Average Standard Limits**
Biochemical Ox Demand (5 Day)	12.5 mg/1	44.4 mg/1	30mg/1
Total Suspended Solids	11,331 lbs/day	38,449 lbs/d	42,012 lbs/d
Fecal Coliform	50/100 ml	650/100 ml	200/100ml
True Color	1,043 std. units	2,035 std. units	50 std unit
2,4,6 Trichlorophenol		< 10 µg/1	
Pentachlorophenol		< 50 µg/1	
Zinc (one sample)		80 µg/1	
Chloroform (w/ plant modification)		238 mg/1	3.3mg/1
2,3,7,8 TCDD (dioxin)		6.61 pg/1	0.014 pg/1
2,3,7,8 TCDF (furans)		5.62 pg/1	

Commodity Specification: Long-Term Health Risks from Dioxin

Response/Causal Agent: Dioxin exposure causes a range of health risks from life-threatening cancers of the soft tissues to nonlife-threatening skin problems, fertility problems. and birth defects.[5] In addition, evidence suggests dioxin can cause immune system suppression in mice at low dose levels, and it is a known promoter of other carcinogens.[6] Dioxin can contaminate the air, water, and soil, and exposure occurs through three possible pathways: inhalation, absorption, or ingestion. Dioxin is more easily absorbed in small doses.

Increasing the exposure levels of dioxin may increase the risk of immunosuppressant health effects,[7] and if accumulated exposure levels increase,[8] the population may have a risk of cancer. Therefore, we may specify our commodity as a particular set of symptoms such as increased disease days from failure of the immune system to fight colds, flu, and other common

[5] See Schmidt (1992).
[6] See Schmidt (1992).
[7] See Schmidt (1992).
[8] Dioxin has a long half-life, causing potential accumulation in the body.

ailments, and as an increase in the risk of chronic illness. We may also specify the commodity as an increased risk of cancer mortality.

Elevated cancer mortality risk is evident in the health statistics for the area. Both Haywood and Cocke Counties have cancer rates greater than the national average (see Table 4). Cancer mortality rates for the two counties range from 7 percent to 35 percent greater than the national average.[9] Chemical workers exposed to dioxin in the U.S. and Germany have been found to have cancer mortality rates 15 percent to 24 percent greater than their national averages for all cancers. In the U.S. those with long-term exposures to dioxin at chemical plants had rates 87 percent above normal in one study and nine times higher than the general population in another.[10]

TABLE 4. AGE-ADJUSTED CANCER MORTALITY RATES (PER 100,000 PERSONS)

Year	Haywood[a]	Cocke	U.S.[b]
1979 - 1981	135.44	141.2	132.0
1982 - 1984	167.89	158.4	133.0
1985 - 1987	179.24	153.7	132.7
1988 - 1990	NA	151.9	133.7

[a]These data are quoted for years 1979 through 1981, 1981 through 1985, 1984 through 1988.
[b]U.S. data are for years 1979, 1981, 1984, and 1989, respectively.

Risk Definition: The policy under consideration (increasing the maximum exposure limits) affects the probability of exposure and thus the probability of immune suppression health effects, as well as the probability of cancer mortality.

Temporal Dimension: Although the immune system effects occur soon after exposure, cancer has a latency period. The immune system problems persist as long as a potent level of the

[9]These figures do not correct for other cancer-causing behaviors and thus cannot be attributed solely to dioxin exposure.
[10]See Schmidt (1992).

chemical remains in the body and thus cause chronic problems given the long half-life of the chemical.[11] The cancers are also chronic.

Voluntary or Involuntary: Exposure to the hazard in our case study is both voluntary and involuntary. Paper mill workers and those who live in the communities surrounding the mill voluntarily expose themselves to the hazard, assuming they are aware of the chemical's presence.[12] Although we recognize their relative ability to relocate, downstream residents are involuntarily exposed.

Exposure Levels: The policy site population has been exposed to low dose levels for long time periods. Present exposure levels for the communities surrounding the mill and the downstream communities are considered low. Mill workers, however, may have higher exposure levels. A July 1989 EPA Fact Sheet (EPA, 1988) on the Pigeon River in North Carolina reported dioxin levels in fish fillet samples of 2.3 to 80 ppt and wholefish levels of 36 to 91 ppt. In Tennessee they found 0.17 to 29.3 ppt in fillets.[13] The NC state limit for dioxin is 0.014 pg/l or 14 ppt.

Both states have given advisories against eating fish from the Pigeon River, and neither state has classified the river for use as domestic water supply. Residents along the river or users of the river have had a lifetime of exposure if they have any regular contact with the river, for example, through recreational activities such as fishing and boating or through drinking from contaminated wells. Tests performed in 1987 by the Tennessee Health Department found toxins, such as furans, contaminating wells of Hartford residents.

Policy Site and Sample Characteristics

Socioeconomic: Both Cocke and Haywood Counties are rural areas. Table 5 summarizes the 1990 demographic data for these two counties.

Location and Temporal: Both government and private insurance programs are available to consumers in both counties; medical care is similar to that available in rural areas in the U.S. Exposure has occurred over a period of 80 years, the time frame in which the

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[11] All references to immune system problems are presently hypothetical because evidence of this health effect has only been found in mice.

[12] This assumes these households can afford to move and choose not to. The costs of moving could be seen as conservative estimates for benefits of health risk reductions. See averting behavior literature (Abdalla, Roach, and Epp, 1992)

[13] The higher levels were obtained from the whole body of a bottom-feeding white sucker. Tests of surface-feeding sunfish yielded a dioxin level of 12 ppt. The variation in the levels could be partially explained by food source.

TABLE 5. 1990 DEMOGRAPHIC INFORMATION FOR HAYWOOD COUNTY NC AND COCKE COUNTY, TN

	Haywood County, North Carolina	Cocke County, Tennessee
Population	46,942	29,141
Mean Household Income	$22,698	$17,624
Mean Education	12.1	12 (median)
Male/Female Distribution	47/53	48/52
Racial distribution (W/B)	98/1.4%	97/2.1%
Age Distribution		
>65	18.2%	12.9%
<18	20.8%	24.0%
Median Age	39.9	35.2
Household Size (mean)	2.4	2.58

mill has been operating. Avoidance opportunities are limited but include staying away from the river, not eating the fish, not working at the mill, and moving. Although all of these would reduce exposure, airborne and soil contamination are unavoidable to area residents. Finally, the geographic extent of the market would include those who live in the vicinity of the contaminated portion of the river and in the vicinity of the mill. Given the central location of the river and/or mill in each county and the location of the mill in the two-county area, we can use the county boundaries for the market's geographic definition.[14]

Market and Exchange Mechanisms

Risk Redaction Technology: Dioxin has a half-life of 7 years, giving a long detoxification time frame. Thus, some type of reduction strategy is necessary. Source reduction must occur from either voluntary reduction by industry or government enforcement. Because of the limited number of highly contaminated sites the private sector has little incentive to provide

[14]We include Cocke County in this study because they are directly affected by North Carolina legislation relating to water quality.

the high incineration necessary for cleaning up toxic soils or sludge from rivers and streams. Collective action appears to be the most likely cleanup strategy for source reduction. Individuals can, however, pursue private averting behaviors such as purchasing bottled water or avoiding the river for recreational activities such as swimming and fishing. When transferring values we may consider either collective action or private action values, but the latter may not reflect reduction in the substance from all pathways (i.e., air, water, and soil).

Exchange Medium: The policy site medium would likely be a city water price or taxes, both of which can be applied to a collective reduction strategy.

Nonuse Values and WTP/WTA: Nonuse values are likely present for children, relatives, and possibly others for both the morbidity and mortality impacts. At the policy site, communities have the property right to clean water, but the state is responsible for enforcement of that right. Citizens must convince their government of their preferences; thus we would measure a consumer's WTP to avoid an increase in the dioxin limit (a Hicksian equivalent measure of welfare change).

Benefits Transfer: Valuing the Benefits of Maintaining 14 ppt Limit on Dioxin

In this case study we want to estimate the ex ante economic value to avoid an increase in dioxin limits. Because we have defined our commodity as the probability of morbidity and mortality effects from long-term low dose levels of exposure, we are estimating the value of avoiding an increase in the probability of chronic morbidity or cancer mortality, or both.

A significant amount of research estimates economic values for a reduction in the risk of morbidity or mortality (Gegax, Gerking, and Schulze, 1991; Gerking and Stanley, 1986; Smith and Desvousges, 1987; Viscussi, Magat, and Huber, 1991). Other studies, such as Berger et al. (1987). provide economic values for symptom-free days. Many of these studies have focused on short-term risks where the time between the cause and effect is immediate (accidental death) and on acute health effects such as burns and coughs. Few studies have looked at the chronic and/or latent health effects characteristic of our policy site. Using the criteria suggested earlier, we selected four studies as potential study sites: Viscusi. Magat, and Huber (1991); Gegax, Gerking, and Schulze (1991); Smith and Desvousges (1987); and Berger et al. (1987).[15] Table 6 summarizes the characteristics of these studies.

[15]Given the shortage of morbidity studies available, the Berger et al. (1987) study was selected although it focuses on short-term morbidity effects.

TABLE 6. POTENTIAL STUDY SITES FOR CASE STUDY

	Viscusi et al., 1991	Gegax et al., 1991	Smith and Desvousges, 1987	Berger et al., 1987
Valuation Issue	Morbidity (Mortality) reducing risk of chronic bronchitis	Mortality Increased risk of accidental death	Mortality reducing risk of death in 30 years	Morbidity value of additional sympton-free days for specific symptoms
Valuation Method	• CVM • Pairwise comparisons risk-risk and risk-cost of living • Interactive computer • WTP	• Market based • Hedonic wage model • Mail • WTA	CVM • Direct WTP question • Person to person • WTP	• CVM • Direct WTP question • Person to person • WTP
Risk Measure	Actual risk	Perceived (# workers/4,000)	Actual risk	NA
Study Site	Greensboro, NC	National	Boston Metro area	Denver & Chicago
Sample Size (usable)	389	737	609	137 (illustrative)
Demographics				
Mean Household Income	$35,000-$37,000	NA	$32,500	NA
Mean Education	14	NA	14	NA
Male/Female Distribution	50/50	NA	3961	NA
Racial Distribution (B/W)	—	NA	97/3	NA
Age (mean)	33	NA	42	NA
% > 65	—	NA	17.2	NA
Household size (mean)	2.7-2.8	NA	2.7	NA
% Household with child < 18	—	NA	36	NA

(continued)

TABLE 6. POTENTIAL STUDY SITES FOR CASE STUDY (CONTINUED)

	Viscusi et al., 1991	Gegax et al., 1991	Smith and Desvousges, 1987	Berger et al., 1987
Sensitivity of valuation estimate to individual Characteristics	Yes	No	Yes	No
Valuation estimates	$ per 1/100,00 decrease in risk of chronic bronchitis	Marginal value of safety for workers	See Table 2, p. 100 of Smith and Desvousges	Mean Daily Consumer Surplus
Mean Median	8.83 4.57 Implicit $ value per chronic bronchitis	Union-Blue $2,103,120 Union $1,180,304 All blue $1,180,304	Examples: $ per 5/50 dec. in exposure with combination end pt. risk of 1/100	Cough $75.98 Sinuses $27.32 Throat $43.92
Mean Median	883,000 457,000 $Value per $1/100,000 decrease risk of accidental death	All blue $1,180,304	Mean $14.19 Med. $10.00 End pt. 1/200	Eyes $48.48 Drowsiness $142.00 Headaches $108.71
Mean Median	81.84 22.86		Mean $26.20 Med. $10.00	Nausea $47.88
Mean value of Statistical Life (millions of $)	8.184	1.62	NA	NA

Both Viscusi, Magat, and Huber (1991) and Berger, Blomquist, Kenkel, and Tolley (1987) are CVM morbidity studies, while the Gegax, Gerking, and Schulze (1991) and Smith and Desvousges (1987) are mortality studies. Note Gegax, Gerking, and Schulze is a hedonic wage study and Smith and Desvousges is a CVM study. The Gegax, Gerking, and Schulze study measures WTA for an increase in perceived risk of accidental death. The other studies measure WTP for decreases in the health risks (Viscusi, Magat, and Huber and Smith and Desvousges) and WTP to get an increase in symptom-free days (Berger, Blomquist, Kenkel, and Tolley). Which study should we use for benefits transfer?

Study Selection: Given the specification of our commodity we choose Viscusi, Magat, and Huber (1991) and Smith and Desvousges (1987) as our possible studies. Both studies value chronic or latent health effects, which are similar to the same effects from dioxin exposure.[16] Smith and Desvousges (a mortality study) and Viscusi, Magat, and Huber (a morbidity study) provide demographic information and a sensitivity analysis of their results. Both also value a change in probability not severity. Table 7 compares the Viscusi, Magat, and Huber and Smith and Desvousges study sites with our policy site.

Although several characteristics of the study sites make them appealing for a benefits transfer, the sites also have several important problems. First, a critical problem is the difference in the direction of change for the study sites and our policy site. Viscusi, Magat, and Huber (1991) looks at risk decreases; Smith and Desvousges (1987) look at both increase and decreases. Smith and Desvousges find that consumer values are higher for WTP to decrease risk than WTP to avoid an increase.

If we agree with their findings, we can consider the study sites as upper bound estimates. Second, the policy site includes both chronic morbidity and latent mortality effects, while the study sites include only one or the other. As recommended above, mortality figures may he considered lower bounds. Therefore, we might consider the economic values from both study sites as upper bounds but also consider the Smith and Desvousges (1987) study values as lower bounds. The transfer is imprecise because no benefit estimate applies perfectly. The analyst must now recall the purpose for the estimate and determine the need for accuracy. Finally, must we adjust for the demographic differences in education and income levels at the policy site?

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[16] Smith and Desvousges (1987) study probability of death from exposure to hazardous wastes, and Viscusi, Magat, and Huber (1991) study severe chronic bronchitis. Although chronic bronchitis may not be a specific effect from dioxin exposure, many of the symptoms may be similar but less severe.

TABLE 7. COMPARISON OF POLICY SITE TO STUDY SITE

	Viscusi et al., 1991	Smith and Desvousges, 1987	Policy Site	
Morbidity/Mortality	Morbidity	Mortality	Morbidity and Morality	
Risk def.	• Probability of chronic bronchitis (↓) (-1/100,000)	• Probability of exposure (↓) (-0.05/50 ... -5/50)	• Probability (↑)	
Temporal dimensions	Chronic Illness (serious)	Latent Effect (serious)	Chronic and Latent (mild-serious) (serious)	
Voluntary/involuntary	Involuntary	Involuntary	Involuntary/Voluntary	
Exposure pathway	Air not specified	Ingestion air	Air, Soil, Ingestion	
Exposure level	Low	Variable	Low	
Socioeconomic			A	B
Household income (mean)	$35,000 - $37,000	$32,500	$22,700	$17,500
Years education	14[a]	14[b]	12[a]	12[b]
Male/female distribution	50/50	39/61	47/53	48/52
Racial distribution W/B	—	97/3	98/1.4	97/2.1
% > 65	—	17.2	18.2	12.9
Household size	2.71	2.7	2.4	2.58
% Households with children < 18	—	36	30	38
Exchange Mech.	Paired Comparisons	Taxes Prices	Taxes or Utility Prices	
Reduct. Tech.	Private	Collective	Collective	
Nonuse	No	No	Yes	
WTP/WTA	WTP	WTP	WTP	

[a]Mean
[b]Median

Value Transfer: We must determine whether we are transferring an equation or a specific estimate from the study sites. Whether we use an equation or a specific estimate depends primarily on the information available from the study sites. If an equation and the relevant data are available from our policy site, transfer of an equation would be the preferred route.

In our particular case, a transfer equation exists for the Smith and Desvousges (1987) study for the risk increase case.[17] Given specific exposure and conditional risk levels,[18] age, income, the number of children in a household, and attitudes to hazardous wastes for the policy site population, a researcher can calculate an estimate for policy site WTP to avoid a probability increase. A transfer function is not readily available in the Viscusi, Magat, and Huber (1991) paper.[19]

If a transfer equation is not available, a specific estimate can be used. Our study values would depend on the dose response for dioxin, which establishes the relationship between proposed policy and probability of the health effect. This relationship can be used to determine the appropriate risk change for analysis.

The Smith and Desvousges (1987) mean values for WTP to avoid a 1/100,000 end point death risk increase range from $17.71 to $47.47. The Viscusi, Magat, and Huber (1991) observation values range from $1.50 to $80.00 per 1/100,000 decrease in probability of chronic bronchitis, with a mean of $8.83. The Smith and Desvousges and Viscusi, Magat, and Huber probability levels are significantly different with Smith and Desvousges levels ranging from conditional probability of death of 1/10 to 1/300. Given that these two studies use different approaches and our concerns for double counting raised earlier, these values should be neither compared nor added together.

Recall that both studies' estimates may be considered upper bounds. The Smith and Desvousges (1987) study uses probabilities higher than those we might expect for dioxin, the Viscusi, Magat, and Huber (1991) study is valuing acute morbidity effects that may be more severe than the acute effects expected from dioxin exposure, and Viscusi, Magat, and Huber measures values for risk reduction. In both studies the demographics may also suggest higher values for the study sites due to higher levels of income and education.

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[17] See Smith and Desvousges (1987), page 103.

[18] Conditional risk is the conditional probability of death occurring scaled by 1,000. The risk of exposure is also scaled by 1,000.

[19] Viscusi, Magat, and Huber (1991) do a sensitivity analysis but find the various demographic variables are insignificant. A transfer function is probably available direcdy from the authors.

The actual choice of a WTP figure must depend on the researcher's policy needs. If only rough estimates are required, the above studies may provide adequate guesses. However if more precise measures are needed, researchers may wish to conduct an original benefit estimation study.

CONCLUSIONS, LIMITATIONS, AND FUTURE RESEARCH NEEDS

Benefits transfer is significantly more difficult to apply than to discuss in theory. The most important limitation is the difficulty in finding reasonably similar commodity specification between the new policy and old study sites. The variation across studies in commodity specification makes transfers difficult. To ease this problem we suggested assuming the causal agent does not matter. However, in our study the variation in direction and magnitude of probability change, the severity of health effects, and the appropriate welfare measure posed significant challenges for transfer. Exacerbating this problem is the singular focus of studies on either morbidity or mortality. Although most long-term health risks from environmental substances include both categories of health risks, the relationship between them has not been examined in the literature. Aggregation through the independent valuation and summation of mortality and morbidity impacts may introduce a systematic bias in estimates (Hoehn and Randall, 1989). This topic is important for future research.

After the above limitations have been adequately addressed, we can then turn our research focus to the relationships between the demographic, location, and temporal variables to value estimates. Further research might also include more studies in developing nations to enhance our understanding of demographic and cultural variables on economic values and our potential for international transfers. In addition, the role of prior information on values and Baysian exchangeability should be studied in more detail (Atkinson, Crocker, and Shogren. 1992). The importance for benefits transfer of documentation and presentation of demand equations cannot be overstated. A collective effort to organize existing studies and databases is needed to enhance researchers' ability to conduct transfers.

Further study of disease attributes, causes, and source as they relate to values is warranted. Can we use hedonic methods to evaluate the relationship between disease attributes and values? Finally, researchers' have not exhausted the various questions surrounding valuation methodology as applied to health risk values nor the potential for nonuse values.

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RECREATIONAL FISHING VALUATION: ACID RAIN PROVISIONS OF THE CLEAN AIR ACT AMENDMENTS

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ABSTRACT

Our work group developed a research protocol to assess the likely magnitude of the economic benefits of improved or nondegraded recreational fishing that are expected to result from implementing the Clean Air Act Amendments of 1990. We used data for the study site from the 1990 NAPAP Integrated Assessment, which includes Maine, New Hampshire, Vermont, and New York. The policy site includes Pennsylvania, Virginia, West Virginia, Maryland, New Jersey, and Delaware.

Congress mandated in §812 of the Clean Air Act Amendments of 1990 (CAAA) that EPA conduct a comprehensive analysis of the impact of the CAAA on the U.S. economy, public health, and the environment. This analysis is to include costs, benefits, and other effects associated with compliance with each standard issued for emissions of sulfur dioxide (SO_2) and nitrogen oxides. Title IV of the CAAA mandates a reduction in SO_2 emissions of 10 million tons per year, with a national cap on SO_2 taking effect in the year 2000.

With the reduction in these precursors to acidic deposition, water quality improvements are expected. A potentially significant source of economic benefits from improved water quality is enhanced recreational fishing. This case study involves developing a research protocol to assess the likely magnitude of the economic benefits of improved or nondegraded recreational fishing that are expected to result from the implementation of the CAAA to control precursors of acidic deposition.

Although substantial improvements (nondegradations) in water chemistry and fish populations may be attributed to the CAAA for three regions of the country (i.e., Adirondack region in New York, Mid-Atlantic Highlands, and Mid-Atlantic Coastal Plains), a preliminary economic assessment has been completed for the Adirondacks only. This area together with

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three other northeastern states (i.e., Maine, New Hampshire, and Vermont) was studied as part of the National Acid Precipitation Assessment Program (NAPAP) and preliminary results were included in the 1990 Integrated Assessment.

At the time of the Assessment, the Adirondacks (and the rest of the Northeast) was the only affected region of the country for which all of the linkages from emissions to fish population declines were established. Therefore, the limited resources for the economic analysis were devoted to assessing damages to the recreational fishery in this region. Finally, the analysis was limited to losses to anglers, and researchers made no attempt to assess any potential nonuse values associated with the changed water chemistry and biota. The contingent valuation method of assessing nonuse values was considered too controversial to survive the NAPAP peer-review process.

The Assessment's National Surface Water Survey (NSWS) encompassed all of the regions of the country thought to suffer adverse water chemistry conditions from acidic deposition. However, the Assessment ascertained that only the Adirondack and Mid-Atlantic regions have potentially high losses in waters suitable for the survival of certain fish populations. The economic losses to recreational fishermen in the Mid-Atlantic regions still need to be assessed. Unfortunately, the linkages from emissions to fish populations are less definitive for the Mid-Atlantic regions, particularly the Coastal Plains, than for the Adirondacks. Moreover, relative to the costs of controlling emissions, the benefits of improved fish habitat and populations are likely to be quite small so that a full-scale original study may not be warranted.

However, two arguments can be made for a less ambitious analysis. First, on a regional scale, the damaged conditions of the fishery may represent a significant loss and a disproportionate burden. Second, recreational fishing damages from fish population losses are but one effect of acidic deposition to be considered along with other damages such as, health effects, impaired visibility, and materials damages. Note that with the probable exception of health effects, each of these effect categories includes uses and nonuse values that are affected by acidic deposition. Therefore, although a full-scale original study of recreational fishing in the Mid-Atlantic region may not be warranted by definitive science or the relative costs and benefits of §812 of the CAAA, a less ambitious assessment of the likely extent of damages is appropriate in this policy context. One of the goals of this benefit transfer research protocol exercise is to describe the extent of analysis required by the policy context.

Consistent with the Assessment, the research protocol described here does not address nonuse values. That topic warrants separate treatment and is beyond our scope.

THE BENEFIT TRANSFER RESEARCH PROTOCOL

A benefit transfer can involve a fairly simple practice such as applying estimates of benefits from one study to an entirely new situation. If multiple, related studies are available, researchers may construct weighted averages of benefit estimates. The original functions that generated the benefit estimates can themselves be transferred, and available data from the policy site can be used in place of the means from the study sites to simulate the models. Ever increasing levels of effort can be directed toward methods of assembling, analyzing, evaluating, combining, and interpreting existing information on how people are affected by a change in conditions, and these methods all qualify as benefit transfers.

In this paper, we develop a benefit transfer protocol for exploiting existing *data* collected in an original study, rather than the values or functions estimated from these data. By having access to the data, researchers are not restricted by the modeling assumptions of the original study. Furthermore, we can consider methods of combining the existing data with data from the policy site.

The four types of data needed in an assessment of recreational fishing benefits are

- behavioral data (e.g.. where do anglers fish and how often?);
- population and angler characteristics (e.g., income, age, tastes, and attitudes);
- site characteristics (e.g., fishing quality, size of the water body, cost of access, geographic distribution of waterbodies by type and in relation to the angling population); and
- policy variables (e.g., fish catch rates, presence of fish species, Acidic Stress Indexes).

Our original data for the study site are from the 1990 NAPAP Integrated Assessment, which includes Maine, New Hampshire, Vermont, and New York (Shankle et al., 1990). The policy site includes the Mid-Atlantic states of Pennsylvania, Virginia, West Virginia, Maryland, New Jersey, and Delaware. The data from the Northeast on recreation behavior, site characteristics, population and angler characteristics, and policy variables, may be used alone or in combination with policy site data on these parameters. Presently, population characteristics are readily available for the Mid-Atlantic regions, and we anticipate the future availability of some policy site data on angler characteristics and recreation behavior (e.g., National Recreation Survey). Site characteristic data exist for the policy site, but accessing these data and linking them with the recreation behavior model is a labor-intensive task. Finally, aggregate data on the range of changes in the policy relevant variables are available in the policy region, but these data may not import well into the recreation behavior models that rely on "site"-specific data.

We develop a benefit transfer research protocol that breaks the analysis down into stages. The progression from one stage to the next is based on a value of information analysis similar to the one presented in Deck and Chestnut (1992) and based on Freeman (1984). The titles for some of the stages of the research protocol have been generalized, however, to accommodate our more encompassing interpretation of the types of analyses that qualify as “transfers.” At each stage of the analysis, we attempt to evaluate the benefits and costs of proceeding to the subsequent stage. We based the decision on the cost of obtaining increments in the quality of benefit information relative to an assessment of how important the quality increment is to the policy context. Finally, in our conclusions we suggest some changes in the way we do empirical research to make benefit transfer practical as well as defensible.

Stage 1 begins with the **Qualitative Assessment** of the economic significance of the damaged recreational fishery. Assuming significant damages have occurred and the policy will result in a reduction in damages, the **Transfer Scoping Analysis** is designed. The purpose of this second stage of the exercise is to assess the availability and relevance of existing information (e.g., studies, reports, databases). The third, or **Benefit Transfer Computation/Estimation,** stage is to determine how best to synthesize, analyze, and otherwise interpret the relevant information to quantify the economic benefits associated with the policy. Here, we attempt to specify and estimate recreational fishing demand models using study site data (i.e., the Northeastern states) alone or in combination with other available data sources. If these data sources are inadequate for providing credible estimates of the recreational fishing benefits of reductions in acidic deposition in the Mid-Atlantic Highlands and Mid-Atlantic Coastal Plains, then, moving to the fourth stage may be necessary. The **Update/Validate** stage involves at least some primary data collection (e.g., a pilot study) and model estimation most likely using procedures for combining data from different sources. The forthcoming National Recreation Survey is described briefly because it may provide relevant, but thin, site-specific data that can be combined with other data to update or validate an existing model. For completeness, the fifth step in the Deck and Chestnut (1992) proposed protocol is an original study. We omit this step because it does not involve a “transfer” at all.

Stage 1: The Qualitative Assessment

The objective of the qualitative assessment is to determine the likely economic significance of the changed condition due to the policy. Two important factors influence any conclusions that can be drawn at this preliminary stage of the analysis. The first relates to the magnitude of the change in the condition of the environment that results from the policy and whether an economically relevant endpoint can be measured. The second involves the sensitivity of economic behavior and/or economic welfare to the change in the measurable endpoint. This latter point includes both the responsiveness of individual agents and the overall number of agents (i.e., extent of the market).

Although these points may appear transparent, most scientific research proceeds while lacking sufficient interaction with economists to ensure that useful endpoints are measured. This criticism applies to the NAPAP in spite of an explicit charge to establish the linkages necessary for relating policy-induced changes in sulfur emissions to policy relevant endpoints. Fortunately, endpoints pertinent to an economic assessment of damages to recreational fishing were measured.

Changes in water chemistry are linked to changes in the viability of certain fish populations through an Acidic Stress Index (ASI), which was developed to reflect the combined effects of pH, aluminum, and calcium on aquatic biota. The ASI logistic models, which were estimated using data from laboratory experiments, predict the probability of larval fish mortality. Separate models were estimated for three species types with varying degrees of sensitivity to acidity: sensitive, of intermediate tolerance, and tolerant. The leap of faith for the scientists involved generalizing these results to the field. First, they constructed lake and stream-specific ASI values using index water chemistry from the NSWS. Second, they compared these constructed ASI values with other sources of information on fish response. Fortunately, these comparisons suggested approximate reference levels for acid-base chemistry considered unsuitable for survival of certain fish populations (see Table 1).

Using the ASI reference levels and given estimates of changes in water chemistry, the scientists could then predict the regional losses in waterbodies suitable for supporting the various fish populations. Figure 1 illustrates the percentage of NSWS lakes and streams unsuitable for two classes of fish species: tolerant and sensitive. Note that not all of the acidic stress is due to acidic deposition; the contribution of acidic deposition varies by region and in some cases is not known (see Table 2). We present the results of the NAPAP investigations below.

Adirondacks

Fourteen percent of the lakes in the NSWS are acidic (i.e., have low acid neutralizing capacity [ANC] and low pH) where the primary cause of acidity is attributable to acidic deposition. Acidification has resulted in loss of fish populations. Sixteen percent of the lakes studied have lost one or more fish populations as a result of acidification. Twelve percent of the potential brook trout lakes in this region are too acidic for survival of brook trout populations. In addition, the Adirondack Lake Survey (ALS) shows that up to 30 percent of small lakes (2 to 10

TABLE 1. ACIDIC STRESS INDEX REFERENCE VALUES FOR FISH POPULATIONS IN NSWS LAKES AND STREAMS

	Acid Stress Index[a]	
Fish Population Status	**Lakes**	**Streams**
Absence of all fish species	Tolerant species ASI > 30	Intermediate ASI > 30
Absence of brook trout	Tolerant species ASI > 10	Sensitive species ASI > 30
Absence of other sport fish, such as smallmouth bass and lake trout	Intermediate species ASI > 80	Not Applicable
Absence of acid-sensitive. species, such as minnows	Sensitive species ASI > 80	Sensitive species ASI > 10
Excessive mortality of acid sensitive anadramous fish in the mid-Atlantic Coastal Plain	Not Applicable	Blueback herring ASI > 50

[a]The laboratory toxicity data used to develop the toxicity models were generated by the University of Wyoming as part of the Lake Acidification and Fisheries (LAF) project sponsored by the Electric Power Research Institute.

Source: National Acid Precipitation Assessment Program (NAPAP). 1990. *1990 Integrated Assessment Report.* Washington, DC. p. 31.

acres) are acidic. A potential concern is with declines in a fishery resource unique to the Adirondack region, native brook trout populations in remote, high-elevation, pristine lakes and streams.

Other New England

Five percent of the NSWS lakes are acidic with about one-half probably due to acidic deposition. Little or no chronic acidification is indicated in the state of Maine. Assessments of the effects of acidification on fish populations are inferred from water chemistry conditions About 2 percent of the potential brook trout habitat is too acidic for survival of brook trout populations and 4 percent of the lakes have water chemistry unsuitable for the survival of other sport fish, such as lake trout or smallmouth bass. Chemical conditions in 6 to 7 percent of the lakes in the region are unsuitable for the survival of many minnow species. Northeast streams were not included in the NSWS, but other information suggests that approximately 1,700 (5,000

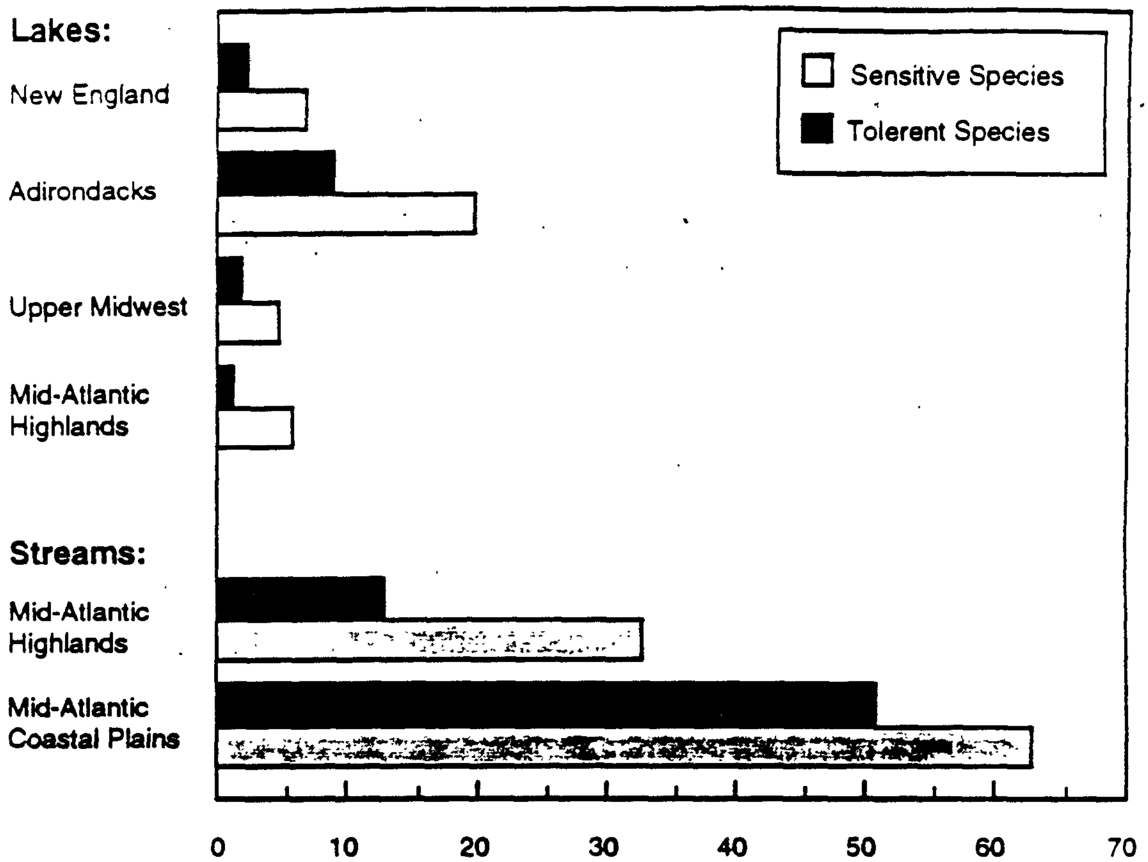

Figure 1. Percentage of NSWS Lakes and Streams Unsuitable for Tolerant and Sensitive Fish Species

Note: NSWS regional lake and stream populations unsuitable, due to acidic stress, for such species as brook trout and sensitive fish specks, such as rainbow trout, minnows, or blueback herring.

Source: National Acid Precipitation Assessment Program (NAPAP). 1990. *1990 Integrated Assessment Report.* Washington. DC. p. 31.

TABLE 2. EFFECTS INFORMATION AND LINKAGES AVAILABLE FROM NAPAP 1990 INTEGRATED ASSESSMENT, BY REGION

Region of Country (by state)	Existing Information/ Models to Estimate Effect of Changes in Deposition on ANC and pH	Existing Information/ Models to Estimate Effect of Changes in pH/ANC on Fish Populations	Expected Effects Due to CAAA
New England + Adirondacks (ME, NH, VT, MA, RI, CT, NY)	Yes	Yes	High
M. Atlantic Highlands (NY, PA, WV, MD, VA)	Yes	Yes	Moderate/High
S. Blue Ridge (GA, SC, NC, VA)	Yes	Yes	Moderate/Low
M. Atlantic Coastal Plains (NJ, DE, MD, VA)	No	Yes	High
Upper Mid-West (MN, WI, MI)	No	Yes	Moderate/Low
Florida	No	No	Low

km) acidic stream reaches exist in this region. This compares with 1,300 acidic upstream reaches in the Mid-Appalachian region. To our knowledge, the effects of stream acidity on fish populations in the Northeast were not investigated by NAPAP.

Mid-Atlantic Highlands (includes the southeastern comer of New York, most of Pennsylvania, and upland portions of Maryland, Virginia, and West Virginia)

Lakes were sampled in only a small part of this region (i.e., Southeastern New York and northeastern Pennsylvania) and 8 percent of the them were acidic. The stream survey covered the Mid-Atlantic region, and 6 percent of the streams were acidic. Chemical composition indicates that atmospheric deposition is the dominant source of acid ions in all the acidic lakes and slightly less than half of the acidic stream length. Data are lacking on the regional status of fish communities in Mid-Atlantic streams, but researchers can draw inferences from the physical and chemical characteristics of the streams when combined with geographical information. An estimated 18 percent of potential brook trout streams (i.e., 37 percent of the National Stream Survey target population) have chemical conditions unsuitable for brook trout survival. Acidity conditions in nearly 30 percent of the streams in the region render them unsuitable for more acid-sensitive species.

Mid-Atlantic Coastal Plain (includes parts of the Piedmont and coastal plain in New Jersey, Delaware, Pennsylvania, Maryland, Virginia, and North Carolina)

Only streams were sampled in this region because lakes are very uncommon. Six percent of streams are acidic, and nearly half the stream length has pH less than or equal to 6.0. Both organic acids and acidic deposition are major sources of acid anions. Unfortunately, numerous factors preclude establishing a causal relationship between acidic deposition and stream acidity, but acidic deposition could be responsible for almost half of current acidification. Indirect evidence does indicate that acidic deposition is a contributor to declines in fisheries and that acidification damages may have been increasing in the last decade. Several important anadromous fish species (e.g., blueback herring) are particularly sensitive to acid stress. Other sensitive anadromous species include striped bass, yellow perch, alewife, American shad, and white perch. Bioassays and models based on bioassays indicate that approximately 60 percent of the coastal streams surrounding the upper Chesapeake Bay in Maryland have a chemical composition during spring baseflow that is toxic for larval anadromous fish. The NSWS chemistry data indicate that acid tolerant fish species in approximately half of the total number of streams in the region may be affected adversely by the acidity. Much of the acid stress is due to acidic deposition; however, field evidence linking fish population declines to acidity or acidic deposition is inconclusive.

Southern Blue Ridge Province (subregion of Southeastern Highlands)

In the Southeastern Highlands, less than one percent of the NSWS stream populations are chronically acidic. Most streams have circumneutral pH (6.5 to 7.0), and fish exhibit little acidic stress under baseflow conditions. The Southern Blue Ridge streams receive sulfur deposition at levels higher than for Adirondack lakes, but the sulfur retention by soils is high and the current stream ANC is relatively high. At present, the number of streams with unsuitable chemistry cannot be modeled because of lack of field data on fish response for the Southern Blue Ridge.

Florida

Although one quarter of the lakes and 39 percent of the streams in Florida are acidic, organic acids and not deposition are the dominant cause. Therefore, acidic deposition is not responsible for the loss of fish populations in this region.

Upper Midwest

Results of the NSWS indicate acidic lakes are 9 percent of the lake population and both deposition and organic acids are contributing factors. The data on the relationship between acid deposition and fish populations are inconclusive, but scientists believe that the effect is minimal. Table 2 summarizes the current information available from the NAPAP 1990 Integrated Assessment by region of the country.

An assessment of the effects of acidic deposition on recreational fishing benefits could be limited to the Adirondacks and a few areas in the rest of New England, the Mid-Atlantic Highlands, and the Mid-Atlantic Coastal Plain: This preliminary qualitative assessment verifies that substantive changes in fish populations in these regions are due to acidic deposition. The extent to which the physical endpoint measured by the scientists (i.e., ASI) can be used for the economic assessment remains an issue. The chemical and biological analyses were intended to provide regional estimates of changes in fish populations (i.e., by percentage of target population of rivers and streams and/or lakes that could support certain fish populations) (see Appendix A for a description of the NSWS.) Economic behavior is influenced by the particular affected waterbodies and not by the quantity affected.

In practice, the economic analysis of recreational fishing damages in the Northeast that was included in the Assessment relied on regression analyses that related angler catch rates to lake-specific forecasted values of the ASI (see Appendix B). In turn, the ASI forecasts were obtained from a regression equation that used variables from the angler survey only. This method of linking the change in a physical endpoint to a change in recreation behavior may not be an option for the policy site. Therefore, we identify as one of the critical issues for this case study and for benefit transfer protocols in general the ability to relate the change in the policy region (i.e., Mid-Atlantic) to the behavior of the policy population.

At the qualitative assessment stage of the benefit transfer exercise, researchers can only ask the larger question: "What is the form of the data coming from the scientists and how can we use it together with available data on recreational anglers to bound the problem?" The discussion above provides detailed information on the form of the physical effect data available from

NAPAP, but we need to identify other scientific research conducted by the states. Moreover, until additional information is gathered on recreational fishery, delineating a protocol for using the information is difficult. However, economic theory and empirical evidence do indicate searching for the following types of information an estimate of the number of anglers; indications that the affected species are desired by the angling population; availability of substitute species; distribution of resource impacts relative to the distribution of the population; whether the problem of fish population losses is reduced by stocking programs; and reversibility of the changes.

By the end of the qualitative assessment the valuation problem should be stated clearly. The extent to which the science is capable of linking the policy change to the change in the physical resource should be described qualitatively, if not quantitatively. Finally, the expected magnitude and uncertainty of the economic consequences should be weighed against the use to which the information will be put to guide the next step of the valuation exercise. Let us assume the results of the qualitative assessment support the next level of analysis-the transfer scoping exercise.

Stage 2: The Transfer Scoping Exercise

A successful scoping exercise will accomplish one or more of the following objectives:

- bound possible values for the effects of the policy change;
- screen studies and other available information for inclusion in the more in-depth analysis of existing information (i.e., Stage 3); and
- determine the relative merits of proceeding to Stage 3, skipping directly to the collection of primary data or truncating the analysis at Stage 2.

The ability to construct bounds on the economic magnitude of the recreational fishing damages avoided because of the CAAA would not necessarily preclude the need for more sophisticated analysis. Additional analysis may be required to enhance the credibility of the estimates.

Two crucial assumptions are at the heart of any benefit transfer exercise. The first relates to the resource and the extent to which it is altered by the policy. Any meaningful benefit transfer relies on experience with evaluating similar changes in similar resources in other places and times. The second assumption involves the people affected by the changed resource. Valuations across populations, time, and space will hold information content for the population affected by the current policy only if a common distribution for underlying preferences exists. Benefit transfer also requires that the ranges of the distribution overlap somewhat.

The comparison of the existing studies on recreational fishing, including the NAPAP analysis of the Northeast fishery, with available information on the Mid-Atlantic states will identify the major similarities and differences in the resources, the extent of the change in the fishery to be evaluated, the populations, and fishing behavior. Considerable judgment will aid in ascertaining whether the study region contexts overlap sufficiently with the Mid-Atlantic to provide any credible information. The overall abundance of freshwater fishing sites, species availability, and the distribution of the fishing sites relative to the population centers are all pertinent factors. Additional characteristics that distinguish the policy regions from study regions are important to identify. For example, the Mid-Atlantic fishery involves rivers and streams only, whereas the Northeast offers lake fishing as well. In addition, both regions offer a variety of cold water, warm water, and anadromous species but not exactly the same species at the same levels of abundance. Do distinctions such as these threaten the homogeneity of the spectrum of fishing opportunities across the contexts?

Differences in fishing behavior will be even more difficult to determine and interpret. For example, conversations with state fishery managers may reveal different fishing patterns, but are they due to differences in preferences or constraints? Mid-Atlantic anglers now fish in rivers and streams, but would they fish in lakes, if available? Do anglers form their preferences based, in part, on the relative abundance of certain species and types of water settings? Currently, very little empirical evidence exists to confirm or deny the stability and uniformity of preferences or to determine how the preferences are formed. Therefore, at this stage of the analysis, maintaining the assumption of a common underlying preference structure is necessary. This assumption can be tested empirically using primary data collected at the policy site.

In summary, prior to engaging in involved computations with the existing data on recreational fishing, the transfer scoping exercise assembles and assesses the evidence on the extent of correspondence with the policy context. If the domains of the resources are similar, including the extent of the changes in the resources (e.g., species availability) to be evaluated, then the existing studies have some information content. Also important to the validity of benefit estimates derived from other contexts is the commonality of the preference distribution across people, place, and time. However, at this stage of the analysis, testing this assumption is not usually possible. To determine the appropriate level of research and analysis effort, the results of this preliminary assessment of the information content of related studies should be balanced against the role that the economic assessment will play in informing the policy debate. To proceed to Stage 3 in the present paper, we assume that we have identified promising information sources and that the cost of extracting and manipulating that information is justifiable.

Stage 3: Benefit Computation/Estimation

Overarching generalizations at this stage of the benefit transfer research protocol are necessarily more vague than at the earlier stages because the most advantageous manipulation of the data will depend both on the available information and the objectives of the analysis. However, for the current situation, where study site recreation demand data are available and descriptive of the policy site, we can be more specific. Furthermore, the following stages of the benefit transfer research protocol have wide applicability due to the numerous policies affecting the quality and quantity of recreation demand opportunities and the multitude of existing, if inaccessible, recreation behavior data sources.

Although our review of existing studies and data sources is incomplete, for the present purpose we assume that the study site data (i.e., the NAPAP recreational fishing survey data for Maine, New Hampshire, Vermont, and New York) exhibit characteristics that overlap with the Mid-Atlantic region. That is, the data have information content, and the challenge relates first to extracting it and second to validating it What distinguishes this benefit transfer research protocol from its predecessors, including the others in this proceedings, is the type of data we have to work with. Prior benefit transfers involved manipulating existing benefit estimates or functions from one or more studies; we have the raw data from an existing study. Therefore, our "transfer" benefit estimation method begins at much the same place as an original study.

First, we specify and estimate the study site model as if we were evaluating the policy change at the study site. This original model should be capable of predicting annual fishing participation rates and economic benefits as a function of quality attributes influenced by the policy. Second, we consider various design changes to facilitate transporting the model to the policy site, noting that restricting the simplified model to the intersection of characteristics across sites is neither necessary nor desirable. Third, we determine whether the design changes seriously affect the description of behavior and the welfare estimates in the study region. For example, the model restrictions are tested using log-likelihood ratio statistics, and the resultant welfare point estimates and ranges are compared. Fourth, in the event that the restrictions do compromise the fit of the model, methods for relaxing the restrictions (e.g.. construction of proxy variables) are investigated. Fifth, the simplified study site model is simulated using available data for the policy site. Finally, the model results are extrapolated to the policy population, which may or may not be well defined. If competing study site "best" models (e.g., alternative functional forms, alternative sample selection correction methods or different methods of integrating fishing site selection decisions with fishing participation decisions) exist, this process can be repeated for each of them.

For future consideration, we anticipate several simplifications for increasing the transportability of the study site model to the policy site. Recall that the policy site offers a different configuration and number of fishing opportunities with different quality attributes to a different population. In addition, the nature of the policy site data available for simulating the model is distinguished from the study site data. Therefore, some of the modifications are directed at accommodating differences in the resources and the populations whereas others stem from incompatible data:

One simplification relates to the differing abundance and types of water resources in the study and policy regions. The Northeastern states have a relatively large abundance of lakes and streams, whereas the Mid-Atlantic states have a large quantity of river and stream miles and very few lakes. One suggestion for modifying the study site model is to include the total number of lakes and/or stream miles (e.g., within driving distance of each county) as a shifter of preferences.

A second simplification involves the availability of fish species. The construction of species aggregates for the modified study site model may depend on the aggregates that best characterize the policy site. For example, both regions may support a warm water fishery and a cold water fishery, but the exact species found in each of the regions, or even within regions, may differ quite a bit.

The form of the data on the effect of the policy change suggests a third modification. The science can predict the percentage losses in stream miles that support certain fish species across the entire region, but determining which streams will be lost may not be possible and determining how stream specific catch-rates would change due to the policy would be even more difficult. Therefore, the study site model should be designed to accommodate "threshold" effects, where, for example, entire recreation sites are removed from the choice set.

Fourth, as is detailed elsewhere in this proceedings (see Cameron, 1992). the study site may be estimated after adjusting each observation by the extent to which it represents of the policy population. The objective of this exercise is to eliminate any bias in coefficients that may be due to a nonrepresentative sample. This is just a sampling of potential model modifications.

Imagining how we might simulate the model for the policy region demands additional simplifications to correct for remaining differences in the distribution and characteristics of fishing resources. For example, suppose we choose a representative individual from each of the counties in the policy region. We still must characterize her fishing opportunities and how they are affected by the policy. Clearly policy site information on the characteristics, quantity, and distribution of fishing opportunities (by species types) is needed, and the form of that data will influence the model design for the study region.

This very extensive exercise is undertaken because researchers believe that the underlying preference structure of the populations in the two regions is the same and because the water resources offer similar recreational fishing opportunities. We intend to use the information on preferences to construct the preference structure of the policy population for the policy site. If the underlying preference structure is not the same, this exercise can have no validity. Furthermore, if the resources in the two regions are not similar in substantive ways, then the study site cannot provide information on people's preferences for the policy resource.

Although employing sophisticated econometric tests of the validity of model estimates of the analysis is not generally possible, assessing the credibility and reasonableness of the estimates using informed judgment is important. The process of gathering the available data on the policy site resource and simplifying and then simulating the study site model forces us to directly address the differences in the recreational fishing resource. However, unless original recreational fishing behavior data are obtained for the policy site, the identical preferences assumption must be maintained. Focus groups involving members of the target population and discussions with officials charged with managing the resource may provide a useful credibility check, but econometric tests require additional data from the policy site. Data on recreational fishing behavior at the policy site, even if insufficient for validation purposes, can nonetheless be combined with existing data sources to update the model and hence increase the credibility of the estimates. Next, we provide a preliminary discussion of the Update/Validate stage of our benefit transfer research protocol.

Stage 4: Update/Validate

This last stage of the benefit transfer research protocol serves the dual purposes of improving and validating the benefit estimates. As with the previous stages of analysis, the extra effort required for increasing the credibility of results will not be necessary or desirable for all policy contexts. However, the more accessible are primary data for the policy site and the methods to manipulate these data, the more attractive and practical utilizing them becomes. Perhaps the first method of obtaining original policy site data that comes to mind is to conduct a pilot study or focus group that is tailored to the current policy context. An alternative approach is to take advantage of national surveys (e.g., National Survey of Fishing. Hunting, and Wildlife Associated Recreation) and secondary data sources (e.g., U.S. Census) that may have been designed with multiple objectives in mind. Finally, a hybrid of the first two data sources is to utilize survey data that were intended to support water-based recreation benefit transfers. One such hybrid is the forthcoming National Recreation Survey.

Devising the methodology for updating or validating the benefit transfer model is beyond the scope of his paper. Indeed, the benefit transfer demand model itself is not provided either. For further details, the reader may consult Parsons and Kealy (1992) who analyze the viability of transferring model estimates for lake recreation choices of Wisconsin residents to the urban sub-population of Milwaukee County. Although she does not include an empirical application, Cameron (in this proceedings) addresses state-of-the-art methodological and empirical issues involving updating and validating empirical models. Here, we describe briefly the National Recreation Survey, which is intended to provide data to support a wide range of methods for updating or validating recreation demand models for benefits transfer.

The Water-Based Recreation component of the National Recreation Survey (NRS) is intended to be administered to a population-weighted random sample of about 14,000 people in the U.S. over the course of a year beginning winter 1993. In addition to obtaining demographic characteristics the phone survey will request information on the total number of trips taken for each of the following primary purposes:

- to fish,
- to boat,
- to swim outdoors (in something other than a pool), and
- to otherwise recreate in a water setting.

The survey will ask for the breakdown between the number of day trips versus the number of trips that included at least one overnight stay. Then, for each of these classes of primary purpose trips, the survey will include a last trip profile. Each “profile” will ascertain destination information (i.e., name of the water body; closest city or town to the waterside; type of water body, that is, lake, stream, wetland or ocean\bay; and additional information needed to construct the travel cost and travel time variables). Pertinent to this case study, the fishing trip profiles will distinguish between types of fishing (i.e., cold water, warm water, or saltwater) and whether the angler used a boat to fish.

Depending on fishing participation rates and how they are allocated between saltwater and freshwater destinations, this survey can lead to a small sample of recreational fishing trips in the regions affected by acidic deposition. The data will also support predictions of the relevant fishing population. Finally, in addition to using these data in combination with existing data sources, the population weights from these data may be useful for weighting the observations on samples from outside of the policy region (see Cameron, 1992).

The sample size may or may not be sufficient to support econometric tests of the validity of parameter estimates for each and every application of the data. Then sensitivity analysis must reveal whether the magnitude of the benefit estimates is sensitive to ranges in the values for the suspect parameters. Conceivably even after such a detailed benefit transfer exercise; the policy context may dictate an original study.

RESEARCH NEEDS

This exercise in designing a benefit transfer research protocol highlights the need for several changes in the direction of applied research. First, benefit transfers do not eliminate the need for data; rather, their success depends on researchers adopting practices to ensure the more efficient utilization of existing data. In particular, no matter what research objective is pursued, data sharing should become one of the goals of the research. Second, research on methods of combining information is needed and should both reflect the type of data that have been collected in the past and influence how data are collected in the future. Third, emphasis on statistical significance should not overshadow emphasis on factors that influence the magnitude of benefit estimates. This suggests that research on reducing uncertainty should focus on the factors that most affect the magnitude of the estimates. Meta-analysis may be particularly useful for identifying those factors. Finally, methods of quantifying the uncertainty of the estimates are needed. In general, research to substantiate the scientific basis for conducting benefit assessments will "transfer" directly to improved benefit transfers.

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