ESSAYS ON AGRICULTURAL EXTERNALITIES AND BENEFIT TRANSFER OF RECREATIONAL FISHING VALUE

DISSERTATION

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By

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ABSTRACT

This dissertation explores the physical and economic aspects of agricultural externalities and the economic value of marine and stream recreational fishing. The effects of pesticide use and tillage practice on water treatment costs and pesticide contamination in treated water are empirically investigated in the first essay. The economic value of marine recreational fishing, the value of access to fishing sites and fishing quality improvements, is examined in the second essay. The third essay examines the economic value of stream recreational fishing: the value of fishing trips and water quality improvements. To evaluate the value of recreational fishing, both second and third essays use *benefit transfer* techniques.

The first essay, *Empirical Investigation of Agricultural Externalities: Effects of Pesticide Use and Tillage System on Surface Water Quality and Treatment Costs*, focuses on the off-farm water quality and water treatment cost effects of upstream and nearby agricultural practices (pesticide use and tillage system), specifically on the pesticide contamination in finished public surface water and water treatment costs in the Maumee River Basin, a major tributary to Lake Erie, located in northwestern Ohio, northeastern Indiana, and southeastern Michigan. Pesticide contamination level in treated surface water and average chemical cost of treating surface water are related to farming

practices and other environmental variables. Findings indicate significant relationships between farming practices and both surface water quality and treatment costs. Average chemical cost per million gallons of water decreases by 1.95% for a 1% reduction in pesticide application, while pesticide contamination level decreases by 4.32% for a 1% more adoption of conservation tillage in a typical watershed area in the Maumee River Basin.

The second essay, *The Economic Value of Marine Recreational Fishing: Applying Benefit Transfer to Marine Recreational Fisheries Statistics Survey (MRFSS)*, conducts a comprehensive survey of benefit transfer techniques including historical background, methodologies, and procedures. Then, benefit transfer technique is applied to the estimation of access value to fishing sites and willingness to pay for better fishing experience in a marine recreational fishing environment of the coastal states in the Northeast and Southeast U.S. Using 1994 Northeast and 1997 Southeast MRFSS data, benefit transfer estimates are compared with original value estimates to empirically examine the validity of benefit transfer. Although benefit transfer error could go up to over 400% of original estimates for some cases, the magnitude of benefit transfer error is less than 100% of original estimates for most cases. Since two data sets used for benefit transfer exercise are from different regions and years, whether regional or temporal variation is more responsible for benefit transfer error can not be determined with current data.

The third essay, *Recreational Fishing Value Estimation of Water Quality Improvements in Western Ohio Using Benefit Transfer*, presents methods for estimating the value of recreational fishing trips and water quality improvements in two watersheds supporting a warm freshwater recreational fishery, the Stillwater River Watershed and Maumee River Basin, in western Ohio using benefit transfer. These two watersheds are further disaggregated into several local stream segments within the watersheds to provide regional results for larger watersheds and to help local policy makers target their efforts more efficiently and effectively. Findings are that annual recreational fishing benefits of water quality improvements are $2,255,616 ($2,759,225 or $3,966,716) and $6,236,853 ($5,395,609 or $7,171,617) with about $44 ($54 or $77) and $58 ($50 or $66) of annual per angler benefits using average value transfer (two function transfer) estimates in the Stillwater River Watershed and Maumee River Basin respectively. These estimates along with disaggregated results in terms of local stream segments and angler types could serve as an initial set of approximated recreational benefits of any local environmental policy involving water quality improvement in inland streams and rivers, at least in terms of recreational fishing.

The measurements of both agricultural externalities and recreational fishing value can be used to help policy makers manage available resources more efficiently and effectively in administrating conservation and/or environmental programs. As is always the case with any non-market valuation technique, careful professional judgments and efforts should be practiced before adopting externality measurements of agricultural practices and benefit transfer estimates of any recreation activity at any stage of policy formulation.

Dedicated to my wife and parents

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FIELDS OF STUDY

Major Field: Environmental and Resource Economics

Minor Fields: Econometrics, Industrial Organization, and Labor Economics

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ESSAY 1

EMPIRICAL INVESTIGATION OF AGRICULTURAL EXTERNALITIES: EFFECTS OF PESTICIDE USE AND TILLAGE SYSTEM ON SURFACE WATER QUALITY AND TREATMENT COSTS

ABSTRACT

This essay examines externalities from agricultural practices in the Maumee River Basin, a major tributary to Lake Erie. Pesticide contamination level in surface water and average chemical cost of treating water are related to farming practices and other environmental variables. Findings indicate significant relationships between farming practices and both surface water quality and treatment costs: average chemical cost per million gallons decreases by 1.95% for a 1% reduction in pesticide application, while pesticide contamination level decreases by 4.32% for a 1% more adoption of conservation tillage in a typical watershed area in the Maumee River Basin. However, it is possible that there is a conflict between more conservation tillage and less pesticide use since conservation tillage may have to rely more on pesticides for weed, insect, and fungal controls. If the relationship between communities' health-related costs and pesticide concentrations in water could be identified, these results would be more meaningful for policy recommendations.

1.1 Introduction

Sources of water resource pollution could be broadly classified into point sources and non-point sources although the distinction between them may sometimes be ambiguous. Point sources are most readily identified with industrial sources such as manufacturing, processing, power generation, and waste treatment facilities where pollutants are delivered through a pipe (a specific and identifiable discharge point), while non-point sources include areas such as agricultural crop fields, parking lots, and golf courses (many different unidentifiable polluters) (Ritter and Shirmohammadi 2001). Non-point source (NPS) pollutants such as sediments, nutrients, pesticides, and pathogens could be transported either to surface water by precipitation (rainfall or snowmelt) or irrigation runoff across the land surface or to ground water by percolating water through the soil. Therefore, the magnitude of NPS pollution is greatly influenced by the volumes of water runoff and percolation that are, in turn, dependent on climate factors and site-specific land characteristics such as soil characteristics (e.g., erosion factor), land management (e.g., agricultural practices), and topography (e.g., slope).

Among many sources of NPS pollution, the four leading sources of NPS pollution in Ohio are agriculture, physical changes to stream channels (hydro modification), mining, and urban runoff (The Ohio State University Extension). According to the U.S. Environmental Protection Agency (USEPA), agriculture is the leading cause of impaired water quality in rivers and lakes, and it is also among the leading causes of impaired estuaries and shorelines in the U.S. Potential adverse effects of agricultural practices on the quality of the Nation's drinking and recreational water resources have become a big concern since agricultural NPS pollutants are now the single largest contributor to the Nation's surface water quality problem. This study focuses on the off-farm water quality and water treatment cost effects of upstream and nearby agricultural practices, specifically on the pesticide contamination in finished public surface water and water treatment costs in the Maumee River Basin, a major tributary to Lake Erie, located in northwestern Ohio, northeastern Indiana, and southeastern Michigan.

Agricultural NPS pollution could be caused by suspended sediments, nutrients (nitrogen and phosphorus), pesticides, pathogens, and salts that are carried by rainfall, snowmelt, or irrigation water into both surface and ground waters. Water runoff from cropland carries these NPS pollutants into surface water, degrading public drinking water supplies, impairing the quality of commercial and recreational water resources, and damaging aquatic ecosystems and wildlife. Pesticides and other agricultural chemicals applied to cropland could also enter aquifers containing ground water that may be used for drinking water, imposing risks to human and animal health. This study tries to empirically assess both physical and economic consequences of farming practices on nearby and downstream communities. Direct physical effect is measured by the pesticide concentration level in the finished public surface water systems (*water quality effect*), while economic consequence is measured by water treatment costs (*water treatment cost effect*) in nearby and downstream communities. Both water quality and water treatment cost effects are assumed not to be taken into consideration when farmers make their farm management decisions such as the rate and timing of pesticide application and the choice of tillage system unless they have proper economic incentives or legal/policy requirements.

In this sense, farming practices may have undesirable externality effects on surface water quality and water treatment costs as well as possible impacts on aquatic ecosystems and wildlife in nearby and downstream communities.

By quantitatively evaluating water quality and water treatment cost effects of agricultural practices, we could compare, at least in terms of these externalities, alternative agricultural or environmental policies that may promote different farm management practices (e.g., pesticide application and tillage system) depending on the incentive structure of each policy. Internalizing these externalities by giving farmers proper economic incentives to take these effects into their consideration should be carefully considered in the process of formulating many agricultural or environmental policies and programs.

1.2 Pesticides and Water Quality

Currently, there are more than 30 classes of registered pesticides including herbicides, insecticides, and fungicides for weed, insect, and fungal controls respectively. On-farm pesticide use was about 401 million pounds in the mid 1960s, and pesticide use more than doubled by 1980 to nearly 851 million pounds. Since the mid 1980s, total pesticide consumption has increased only moderately to 906 million pounds in1996 (Ritter and Shirmohammadi 2001). Since the primary purpose of pesticides is to control weeds, insect pests, and fungus to improve the quality and quantity of agricultural products from cropland, a productivity effect of pesticides on farmers and consumers has been a main concern of many existing economic studies. For example, both Ribaudo and Bouzaher

(1994) and Fernandez-Cornejo, Jans, and Smith (1998) focus on economic effects of reducing pesticide application on producers and consumers of agricultural products.

Besides a positive effect of agricultural pesticides on crop yields, there are also significant negative effects, usually not accounted for by farmers, on the quality of surface and ground waters. Pesticide residues reaching surface water systems through runoff water may harm fresh water and marine organisms, damaging recreational and commercial fisheries. Ground water is also vulnerable to pesticide contamination with significant geographic variation. Areas with sandy, highly leachable soils and high application rates of toxic or persistent pesticides are generally highly vulnerable to pesticide contamination in ground water. Consumption of surface or ground water that is contaminated with pesticides may impose significant risks to human health, depending on the amount of pesticides that people ingest and the duration of exposure. In addition to undesirable effects on human health, pesticide contamination in surface and ground waters may also adversely affect aquatic ecosystems and wild life. Commonly found agricultural pesticides in surface and ground waters include atrazine, metolachlor, and alachlor.

1.2.1 The Fate and Transport of Pesticides

The environmental fates of pesticides applied to cropland are illustrated in Figure1.1. When a pesticide reaches the soil, it may be absorbed by the plant, destroyed by degradation processes, attached (adsorbed) to soil particles, or leached down through the soil.

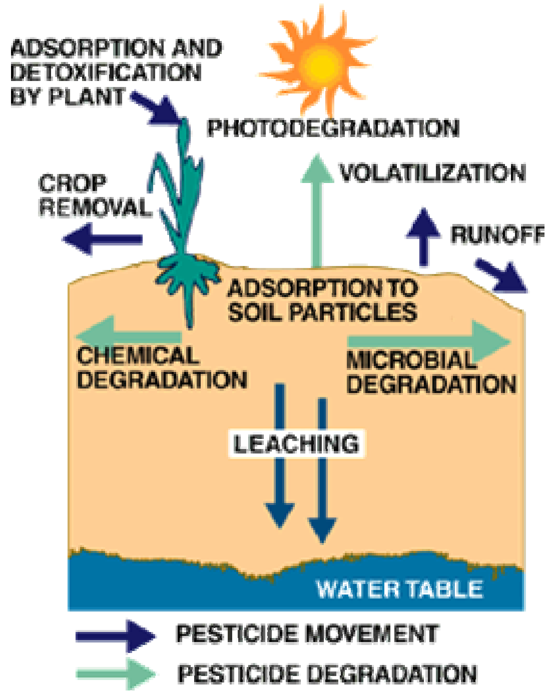

Figure 1.1: Environmental Fates of Pesticides
Source: Ramsay, Cogger, and MacConnell (1991)

The pesticide's environmental fate and transport process depend on the cumulative effects of pesticide (e.g., adsorptivity, solubility, volatility, and degradation rate) and soil (e.g., soil texture, organic matter, and erodibility) characteristics, application methods (aerial or ground), and site conditions (e.g., topography, weather, and irrigation). Some portion of pesticides dissolved in runoff water or adsorbed to eroded soil particles may be washed into streams, rivers, lakes, and estuaries eventually impairing the quality of surface water. Pesticides could also reach ground water by leaching through the soil. An increase in the level of pesticide concentration in drinking water supplies (surface or ground water) could cause a significant increase in water treatment costs since public water systems are required to have additional treatment when certain pesticides exceed health-safety levels. Some studies (Forster, Bardos, and Southgate 1987; Dearmont, McCarl, and Tolman 1998; Holmes 1988) suggest that water quality measured in the form of turbidity or sediment loading could actually influence the costs of producing drinking water.

1.3 Tillage and Water Quality

The choice of a tillage system could influence the quality of surface and ground waters in various ways depending on the soil and pesticide characteristics along with the rate, method, and timing of pesticide application. Tillage could influence sediment loading by affecting the soil erosion by water and pesticide contamination in surface and ground waters by affecting the environmental fate and transport of pesticides applied to cropland. Crop residue cover on the soil surface will increase the opportunity time, the time water remains in contact with soil, by impeding and slowing down the flow velocity of runoff water. Increased opportunity time will influence the amount of water, containing dissolved pesticides, infiltrated into the soil that, in turn, determines the amount of runoff water. Residue cover can protect the soil surface from erosion by absorbing raindrop impact (soil particle detachment), reduce surface crusting and sealing, decrease the velocity of runoff water, and increase soil moisture.

The type of tillage system adopted on agricultural cropland could be an important factor influencing the quality of surface water in nearby and downstream communities along with other factors such as pesticide application, climate conditions, topography, and soil and pesticide properties. The amount of pesticides applied to cropland is in fact determined by the type of crop planted, tillage system used, and geographic location. Different tillage systems may require different amounts of pesticide application to cropland, and may also have different effects on the soil erosion rate that could eventually influence the volume of runoff water containing dissolved pesticides. Changes in agricultural practices, such as crop choice and tillage system, and accompanying changes in pesticide application rate to cropland could have significant water quality and water treatment cost effects on the communities outside agricultural commodity markets, let alone direct economic effects on the producer (possible changes in crop yield and net profit) and the consumer (possible change in consumer surplus).

Environmental consequences of farming practices have been addressed by many existing studies by investigating the impact of conservation tillage on the quality of surface water (Baker and Laflen 1983; Fawcett, Christensen, and Tierney 1994; Gaynor, MacTavish, and Findlay 1995; Ghidey and Alberts 1998; Myers, Metzker, and Davis 2000; Forster 2002). Myers, Metzker, and Davis (2000) illustrate that the increased use of conservation tillage corresponds to decreases in the suspended-sediment discharge over time at two locations in the Maumee River Basin. On average, they find that conservation tillage was used on 55.4 percent of all crop fields in the Maumee River Basin during 1993-1998. Forster (2002) summarizes positive effects and unintended environmental consequences of conservation tillage practices using a simulation model. The positive effects of conservation tillage include improved farm profits and the decrease in some agricultural NPS pollutants including sediment, organic nitrogen, and total phosphorus loadings. However, he also finds some unintended environmental consequences such as the increase in some agricultural NPS pollutants including herbicides and nitrates.

1.3.1 Crop Residue Management (CRM)

In contrast to conventional tillage systems that rely on moldboard plow or other intensive tillage operations with little or no management of residue, crop residue management (CRM) systems involve the use of cover crops and other conservation practices that leave sufficient residue to protect the soil surface from water and wind erosions. CRM is a year-round conservation system that usually involves a reduction in the number of trips to the field with tillage implements and in the intensity of tillage operations, including the elimination of plowing (inversion of surface layer of soil), designed to protect soil and water resources and to provide additional environmental benefits. CRM begins with the selection of crops that can produce sufficient amount of residue to reduce the soil erosion by water and wind, and may involve the use of cover crops after the crops producing low residue. Site-specific amounts of residue cover needed are usually expressed in terms of percentage of the soil surface covered, but may also be in terms of pounds. Tillage systems considered under CRM include conservation tillage (no-till, ridge-till, and mulch-till) and reduced tillage. CRM is generally a cost-effective way of improving environment by protecting soil and water resources, and also able to lead to higher farm economic returns (Forster 2002) by reducing fuel, machinery, and labor costs while maintaining or increasing crop yields.

1.3.2 Economic and Environmental Benefits of Conservation Tillage

Conservation tillage refers to any tillage and planting system that covers *30 percent or more* of the soil surface with crop residue, after planting, to reduce soil erosion by water or any system that maintains *at least 1,000 pounds* per acre of flat, small grain residue equivalent on the surface throughout critical wind erosion period where soil erosion by wind is the primary concern. Main factors influencing the amount of crop residue on the soil surface are the type of crop which establishes the initial residue amount, its fragility, and the type of tillage operations prior to and during planting. According to Conservation Technology Information Center (CTIC) National Crop Residue Management Survey, conservation tillage systems include no-till/strip-till, ridge-till, and mulch-till while other tillage types in the survey include reduced-till and conventional or intensive till.

No-till system leaves the soil undisturbed from harvest to planting except for nutrient injection, and is currently the most highly publicized and promoted conservation tillage in Ohio, Midwest, and the U.S. Ridge-till system is similar to no-till in that the soil surface is not disturbed from harvest to planting. Residue is left on the surface between ridges, and a major benefit over no-till seems to be the opportunity for earlier planting on more poorly drained soils. Mulch-till system is a full-width tillage involving one or more tillage trips that disturb the entire soil surface prior to and /or during planting. Reduced-till

(15-30% residue) and intensive-till (less than 15% residue) systems leave less than 30 percent of residue cover or less than 1,000 pounds per acre of small grain residue equivalent throughout critical wind erosion period.

There are some economic benefits to the farmers adopting conservation tillage, and these factors play an important role in the selection of the type of tillage system. Economic benefits to farmers include reduced labor hours due to a decrease in the number of trips to crop fields, reduced machinery wear, and a saving in fuel consumption. These economic benefits together may increase net profit for the farmer adopting conservation tillage system depending on the size of a possible increase in herbicide costs and a crop yield response to conservation tillage. Economic benefits and costs associated with conservation tillage adoption are likely to depend on environmental factors such as soil characteristics, climate, and topography.

There are also several environmental benefits of conservation tillage although farmers may not take these benefits into consideration when they choose the type of tillage system. Conservation tillage can reduce soil erosion by water and wind up to 90 percent compared to an intensively tilled field without residue protection; improve soil tilth making it easier for plants to establish roots due to increased soil particle aggregation (more small soil clumps); and trap soil moisture reducing water evaporation, slowing runoff, and increasing the opportunity for water to infiltrate into the soil. With conservation tillage, organic matter contained in the soil for future crops increases since less carbon that accounts for about a half of organic matter is released to the air, and the release of carbon dioxide into the atmosphere also decreases due to increased carbon in organic matter. The quality of surface and ground waters could be improved since crop residue cover reduces nutrient and pesticide runoff into surface water by helping the soil hold them, and microbes living in the carbon-rich soils quickly degrade pesticides and utilize nutrients protecting ground water quality. Crop residue on the surface also improves air quality due to reduced wind erosion that, in turn, decreases the amount of dust in the air. Finally, crop residue could also increase wildlife by providing shelter and food source for small animals (CTIC).

1.4 Conservation Tillage Trends

Tillage systems adopted in U.S. agriculture since the mid 1990s are presented in Table 1.1. Conservation tillage systems (no-till, ridge-till, and mulch-till) have been adopted on more than 35 percent of the nation's planted cropland acres since 1995. Among three conservation tillage systems, no-till is the most popular tillage system in the year 2002, covering 19.6 percent of the nation's total cropland acres which is 55.3 million acres. Although more than one third of the nation's agricultural cropland adopts conservation tillage system, intensive-till, that leaves the least amount of residue cover among all tillage practices, alone has been adopted more than the sum of all three types of conservation tillage since 1995 except for 1997 and 1998. For instance, intensive-till has been adopted on more than 40 percent of the nation's cropland in 2000 and 2002. Conservation tillage systems in terms of both proportions and cropland acres have been decreasing continuously since 1997 although only by less than 1 percentage point per year or two-year interval.

Tillage System	*1995*	*1996*	*1997*	*1998*	*2000*	*2002*
No-till/Strip-till	40.9	42.9	46	47.8	52.2	55.3
	14.7%	14.8%	15.6%	16.3%	17.6%	19.6%
Ridge-till	3.4	3.4	3.8	3.5	3.3	2.8
	1.2%	1.2%	1.3%	1.2%	1.1%	1.0%
Mulch-till	54.6	57.5	60	57.9	53.5	45
	19.6%	19.8%	20.4%	19.7%	18.0%	16.0%
Conservation Tillage	98.9	103.8	109.8	109.2	109.1	103.1
Subtotal	**35.5%**	**35.8%**	**37.3%**	**37.2%**	**36.7%**	**36.6%**
Reduced-till	70.1	74.8	77.3	78.1	61.3	64.1
(15-30% cover)	25.2%	25.8%	26.2%	26.2%	20.6%	22.8%
Intensive-till	109.7	111.6	107.6	106.1	127.1	114.3
(<15% cover)	39.4%	38.5%	36.5%	36.2%	42.7%	40.6%
All Planted Acres	278.7	290.2	294.7	293.4	297.5	281.4

Table 1.1: Conservation Tillage in the U.S. (Millions of acres)
No-till/Strip-till, Ridge-till, and Mulch-till are all considered forms of Conservation Tillage.
Source: CTIC National Crop Residue Management Survey

To more clearly illustrate the longer-term trend of conservation tillage systems in the U.S., Figure 1.2 shows the percentage of planted acres adopting each conservation tillage system from 1989 to 2002. One very noticeable phenomenon is a continuous increase in no-till system percentage during the whole sample period. No-till system was adopted only on 5.1 percent of the nation’s cropland in 1989; however, nearly 20 percent of the nation’s cropland adopted no-till system in 2002. Mulch-till system has been the dominant type of conservation tillage throughout the whole sample period except for the year 2002. The adoption rate of ridge-till system has remained fairly low for the whole period never exceeding 1.3 percent.

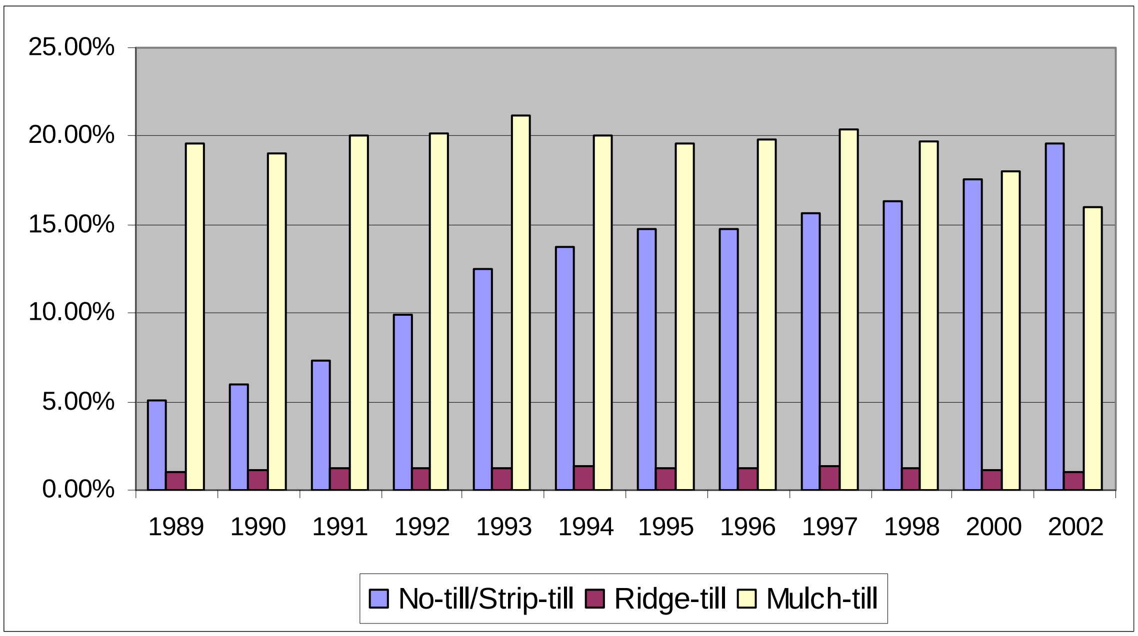

Figure 1.2: Conservation Tillage in the U.S.

Table 1.2 illustrates conservation tillage systems adopted in the whole Midwest region (12 Midwest states) and the three individual Midwest states (Ohio, Indiana, and Michigan) included in the Maumee River Basin area. Similar to the trend shown in the conservation tillage data for the U.S., no-till is the most dominant type of conservation tillage system in the Midwest region for the year 2000 and 2002. Interestingly, no-till system has been the most popular conservation tillage system in Ohio, Indiana, and Michigan since the mid 1990s unlike the U.S. and Midwest region.

In the U.S. and Midwest region, no-till system adoption rate has exceeded mulch-till system adoption rate only in the recent years; in 2002 for the U.S. and in 2000 and 2002 for the Midwest region. The degree of no-till system's dominance over the other types of conservation tillage systems (ridge-till and mulch-till) is noticeably high in Ohio, nearly six times as much as mulch-till system that is the second most popular conservation tillage system. This implies potentially popular practice of no-till conservation tillage system in the Maumee River Basin area that includes 16 counties in Ohio out of 24 counties included in the whole area.

Midwest	*1995*	*1996*	*1997*	*1998*	*2000*	*2002*
No-till/Strip till	17.9%	17.5%	18.7%	19.1%	21.2%	22.2%
Ridge-till	1.7%	1.6%	1.7%	1.5%	1.2%	1.0%
Mulch-till	21.1%	21.4%	22.7%	21.4%	19.0%	17.7%
Conservation Tillage	*40.7%*	*40.4%*	*43.1%*	*42.1%*	*41.4%*	*40.8%*

Ohio	*1995*	*1996*	*1997*	*1998*	*2000*	*2002*
No-till/Strip till	38.1%	36.8%	36.1%	41.6%	45.4%	41.1%
Ridge-till	0.3%	0.3%	0.3%	0.2%	0.2%	0.1%
Mulch-till	8.4%	8.5%	8.5%	7.5%	9.4%	7.3%
Conservation Tillage	*46.8%*	*45.5%*	*44.8%*	*49.3%*	*55.0%*	*48.5%*

Indiana	*1995*	*1996*	*1997*	*1998*	*2000*	*2002*
No-till/Strip till	37.5%	34.0%	33.6%	35.7%	39.5%	38.9%
Ridge-till	0.3%	0.2%	0.2%	0.1%	0.2%	0.2%
Mulch-till	12.5%	9.3%	13.9%	12.5%	12.0%	11.4%
Conservation Tillage	*50.2%*	*43.4%*	*47.7%*	*48.4%*	*51.7%*	*50.5%*

Michigan	*1995*	*1996*	*1997*	*1998*	*2000*	*2002*
No-till/Strip till	24.7%	27.2%	28.0%	27.7%	23.0%	23.4%
Ridge-till	0.1%	0.0%	0.0%	0.0%	0.0%	0.2%
Mulch-till	21.9%	21.8%	21.8%	23.5%	8.6%	7.6%
Conservation Tillage	*46.6%*	*49.1%*	*49.8%*	*51.2%*	*31.6%*	*31.3%*

Table 1.2: Conservation Tillage in the Midwest, OH, IN, and MI
No-till/Strip-till, Ridge-till, and Mulch-till are all considered forms of Conservation Tillage.
Source: CTIC National Crop Residue Management Survey

Tillage System	*1995*	*1996*	*1997*	*1998*	*2000*	*2002*
No-till/Strip-till	3,724.2 43.3%	3,470.1 40.7%	3,381.8 39.2%	3,692.7 44.2%	3,938.1 47.6%	3,184.1 39.9%
Ridge-till	9.4 0.1%	16.1 0.2%	13.8 0.2%	8.9 0.1%	5.9 0.1%	9.1 0.1%
Mulch-till	885.8 10.3%	1,058.6 12.4%	1,048.2 12.2%	845.0 10.1%	934.5 11.3%	637.3 8.0%
Conservation Tillage Subtotal	4,619.3 **53.7%**	4,544.8 **53.3%**	4,443.8 **51.6%**	4,546.6 **54.4%**	4,878.5 **59.0%**	3,830.5 **48.0%**
Reduced-till (15-30% cover)	938.4 10.9%	922.5 10.8%	946.6 11.0%	425.9 5.1%	792.7 9.6%	722.3 9.1%
Intensive-till (<15% cover)	3,049.6 35.4%	3,066.2 35.9%	3,228.4 37.5%	3,384.3 40.5%	2,599.3 31.4%	3,423.3 42.9%
All Planted Acres	8,607.4	8,533.4	8,618.8	8,356.9	8,270.5	7,976.1

Table 1.3: Conservation Tillage in the Maumee River Basin (Thousands of acres)
No-till/Strip-till, Ridge-till, and Mulch-till are all considered forms of Conservation Tillage.
Source: CTIC National Crop Residue Management Survey

The Maumee River Basin area consists of 16 counties in Ohio, six counties in Indiana, and two counties in Michigan. Compared to the national trend in conservation tillage adoption, conservation tillage systems have been much more popular in the Maumee River Basin area since the mid 1990s as shown in Table 1.3. In these years, more than or nearly half of the planted cropland in this area adopted one of the conservation tillage systems. As might be predicted from the trend in Ohio's conservation tillage adoption, no-till is obviously the most widely used conservation tillage system in the Maumee River Basin, being adopted on about 80 percent of the planted cropland with conservation tillage in this area since the mid 1990s. Since the mid 1990s except for the year 2002, no-till has been dominant over intensive-till that has been the most popular tillage system for the period of 1989-2002 in the U.S.

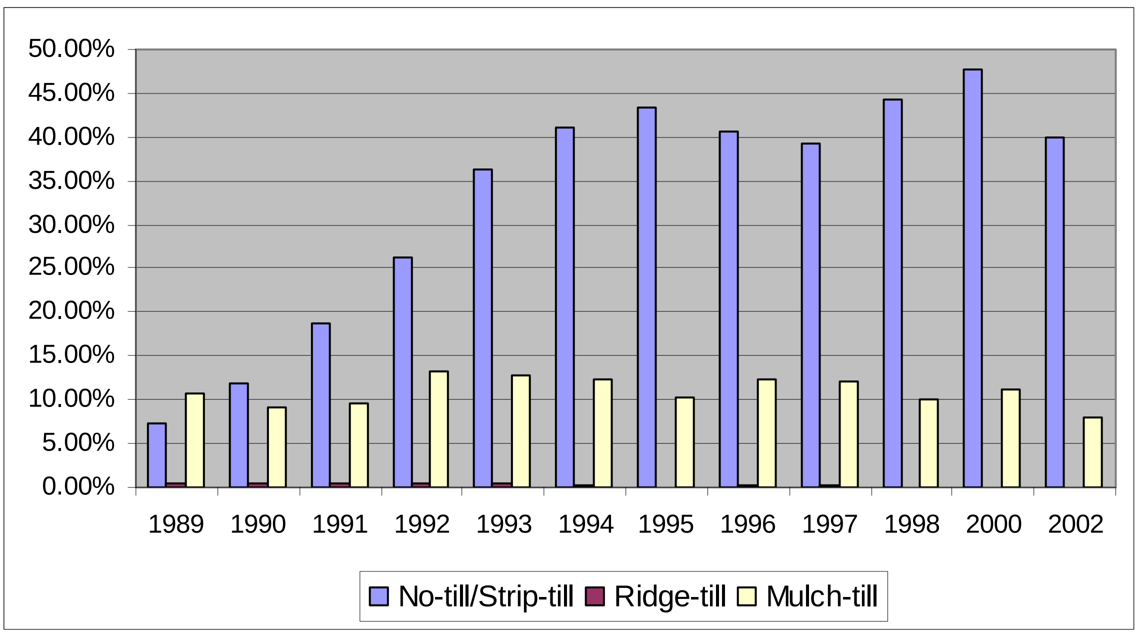

Figure 1.3: Conservation Tillage in the Maumee River Basin

The dominant use of no-till conservation tillage system is more evident in Figure 1.3 which illustrates conservation tillage practice in the Maumee River Basin. The adoption rate of no-till conservation tillage peaked in year 2000 with a continuous increase up to the mid 1990s and an initial decrease followed by a subsequent increase in the late 1990s. Similar to the national trend, conservation ridge-till adoption rate has never exceeded 1 percent of the total planted cropland for the whole period of 1989-2002. One primary concern from the conservation tillage trend in the Maumee River Basin is the recent decrease in all three conservation tillage systems since 2000 in spite of many government programs that attempt to reduce agricultural NPS pollution in this area from the early 1970s to the late 1990s by focusing on voluntary adoption of soil and water conservation practices (Forster and Rausch 2002).

In summary, intensive or conventional tillage system has been most widely practiced in the nation's planted cropland although the adoption rate of no-till conservation tillage system has been constantly growing since the late 1980s. No-till conservation tillage system has been most dominantly practiced in the Maumee River Basin area since the mid 1990s even more dominant than nationally most popular intensive tillage system. On average, about 37 percent and 53 percent of the planted cropland adopted conservation tillage system in the U.S. and Maumee River Basin area respectively during the period of 1995-2002.

1.5 Study Area and Data

The Maumee River Basin is located in northwestern Ohio, northeastern Indiana, and southern Michigan, containing about 4.2 million acres of total watershed area. Seventy six percent of the Maumee River Basin area is occupied by row-crop agriculture, containing about 50 percent of the total cropland draining into Lake Erie from both the United States and Canada (Forster et al. 2000). Intensive agricultural activities contribute significantly to the elevated levels of suspended sediments, fertilizers, and pesticides in the runoff to stream waters in the Maumee River Basin. Since the early 1970s, natural resources conservation programs such as the Conservation Reserve Program, Agricultural Conservation Program (before 1997), and Environmental Quality Incentive Program (1997 and after) have been used to reduce sediment and nutrient discharges in the Maumee River Basin and to improve aquatic habitat for fish and wildlife (Forster and Rausch 2002; Myers, Metzker, and Davis 2000). About 88 percent of human water consumption (public and domestic supply) in the Lake Erie-Lake Saint Clair Drainages, within which the Maumee River Basin is included, comes from surface water sources, while about 12 percent of it comes from ground water sources (Myers et al. 2000).

Eleven watershed areas and corresponding surface-water treatment plants in the Maumee River Basin area are included in the analysis; therefore, basic observation units for this five-year period (1995-1999) study are 11 “watershed areas” (agricultural practices and environmental factors data) and corresponding “water treatment plants” (water quality, treatment cost, and plant characteristics data). All water treatment plants are located in northwestern Ohio as shown in Figure 1.4 although the quality of surface water they treat is also influenced by agricultural activities of some upstream counties located in northeastern Indiana and southern Michigan. To account for the possibility that upstream agricultural practices could influence not only the quality of nearby rivers and streams but also the quality of downstream rivers and streams, 11 watershed areas are allowed to overlap each other. Each watershed area includes corresponding water treatment plant's upstream river and stream segments as well as nearby river and stream segments, and the surface water quality treated by each water treatment plant is assume to be affected by agricultural activities of these nearby and upstream river and stream segments. For instance, the water treatment plant located at Bowling Green, downstream Maumee River, is assumed to be affected by agricultural practices of almost the whole Maumee River Basin area (about 4 million acres).

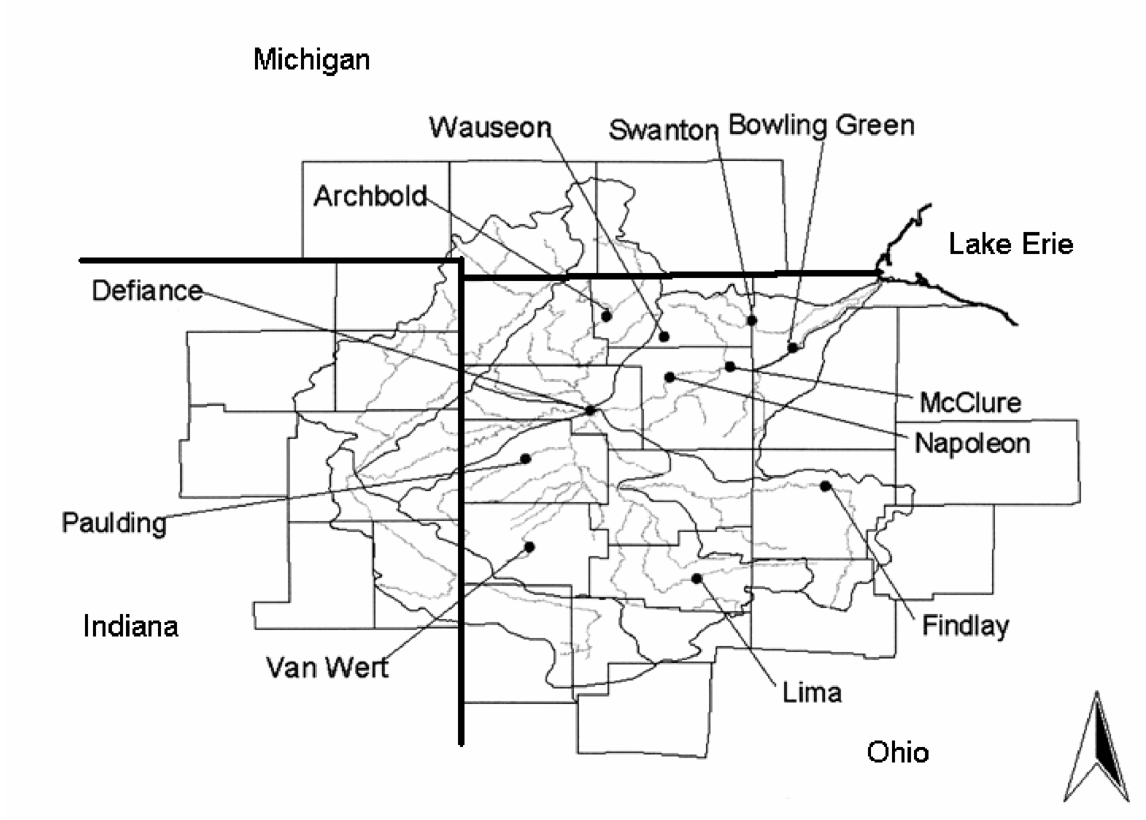

Figure 1.4: Locations of Water Treatment Plants in the Maumee River Basin

Two externality effects, water quality effect and water treatment cost effect, caused by agricultural practices are analyzed in the study. Pesticide concentration level data from the Ohio EPA's Division of Drinking and Ground Waters (DDAGW) and water treatment cost survey data from the 11 water treatment plants in the Maumee River Basin are used to analyze these externality effects. Detailed data on annual costs of treating public surface water from 1995 to 1999 were collected by a mail survey along with plant characteristics data including volumes of water treated and raw water storage capacities. The DDAGW of Ohio EPA conducted a pesticide special study to determine the occurrence of commonly used pesticides (alachlor, atrazine, metolachlor, metribuzin, simazine, cyanazine, and acetochlor) in the finished water of nearly 150 public surface water systems in Ohio.

Agricultural practices considered in this study include the tillage system adopted and the amount of pesticides applied to crop fields. Tillage practice data from 1995 to 1998 were collected through the National Crop Management Survey conducted by CTIC with five different tillage categories including three conservation tillage systems (no-till, ridge-till, and mulch-till) and two other tillage systems (reduced-till and intensive-till). There was no tillage survey in 1999, so the average of 1998 and 2000 data is used to approximate missing 1999 tillage data for the empirical analysis. Pesticide application data are based on the U.S. Department of Agriculture (USDA) National Agricultural Statistics Survey (NASS) Agricultural Chemical Usage, Field Crop Summary data. The county level data on pesticide application are estimated by multiplying the percentage of field crop acres receiving pesticide application, total crop acres in each county, the state average of pesticide application per crop acre, and then summing over all field crops in the county.

Precipitation and soil erosion data are also used to investigate the influence of site-specific environmental conditions on the quality and treatment costs of surface water. Precipitation data were colleted from the Midwestern Regional Climate Center for eight weather stations in the Maumee River Basin. Missing precipitation data for three watershed areas (Archbold, McClure, and Swanton) are substituted by the precipitation data from the closest weather stations. The soil erodibility factor (*K* factor) that quantifies the susceptibility of soil particles to detachment and movement by water as used in the universal soil loss equation (USLE) to calculate soil loss by water was collected from the USDA 1997 National Resources Inventory (NRI).

Finally, all county level data (tillage practice, pesticide application, and *K* factor) are converted into watershed area level data by multiplying them by percentages of total county acres included in each watershed area that constitutes an observation unit in this study, and adding these weighted (weight as the percent of county acres included in the watershed area) county data across all the counties in the watershed area.

1.6 Descriptive Statistics

Some basic characteristics of water treatment plants and corresponding watershed areas along with site-specific environmental factors are summarized in Table 1.4. Eleven water treatment plants vary considerably in terms of their capacities for water treatment and raw water storage. The volume of water treated may be directly related with the number of population served by each water treatment plant. For example, the Lima water treatment plant, serving 74,000 people, annually treats almost 200 times more volume of water than the McClure water treatment plant, serving only 850 people. Noticeably, a considerable part of watershed area contains farmland acres, ranging from 64 percent up to 90 percent with the average of 77 percent of total watershed area. Such a dominance of agricultural usage in the Maumee River Basin area may imply potentially significant effects of agricultural NPS pollutants, including pesticides, fertilizers, and nutrients, on the quality and treatment costs of surface water.

Treatment Plant	*Volumes Treated (mil. gal.)*	*Reservoir Storage (mil. gal.)*	*Pop. Served*	*Total Acres*	*Farmland Acres*	*Prec. (inch)*	*K-Factor*
Archbold	692	300	5,114	222,036	141,723	33.98	0.29
Bowling Green	1,614	170	40,000	3,990,508	3,106,237	32.02	0.33
Defiance	1,539	0	17,000	257,390	179,389	36.55	0.33
Findlay	2,603	6,400	40,000	137,509	112,037	35.35	0.35
Lima	5,167	9,141	74,000	110,799	88,071	38.47	0.36
McClure	26	0	850	3,881,092	3,005,644	30.28	0.33
Napoleon	501	0	10,500	3,647,825	2,803,799	30.28	0.33
Paulding	160	500	3,318	115,398	86,018	34.94	0.33
Swanton	188	110	4,000	18,432	13,962	32.02	0.27
Van Wert	642	780	11,000	14,080	12,730	37.25	0.34
Wauseon	355	350	8,000	4,224	3,198	33.98	0.27
Average	*1,226*	*1,614*	*19,435*	*1,127,208*	*868,437*	*34.10*	*0.32*

Table 1.4: Plant and Watershed Characteristics
Volumes treated and annual precipitations are averages of five-year period (1995-1999). Yearly precipitation data represent annual total amount of precipitation in a watershed area.

Site-specific environmental factors such as precipitation and soil erosion factor also influence the amount of runoff water and thus the quality and/or treatment costs of surface water. The index for soil erodibility factor, K factor, represents soil's inherent susceptibility to erosion, and depends on many site-specific soil characteristics such as infiltration capacity and structural stability. The soil erodibility factor normally varies from near zero to about 0.6 associating lower erodibility indexes with the soils into which water can readily infiltrate. Soils with intermediate infiltration capacity and moderate soil structure stability generally have a K factor of 0.2-0.3, while more easily eroded soils with low infiltration capacities have a K factor of 0.3 or higher. On average, a K factor in the Maumee River Basin area is above 0.3 indicating that soil erosion could be an important consideration for some agricultural or environmental policies that attempt to promote more soil-conserving farming practices such as conservation tillage.

Two environmentally consequential farming practices analyzed in this study, pesticide application and tillage practice, are summarized in Table 1.5. The amount of pesticides applied to cropland may be closely related with farmland acres in a total watershed area. Three watershed areas associated with relatively larger farmland acres, Bowling Green, McClure, and Napoleon, are also the areas with greater amount of pesticides applied to cropland. The relationship between tillage system and pesticide application is not clearly detected at least in Table 5 although conservation tillage systems may depend more heavily on pesticides for weed, insect, and fungal controls. The amount, timing, and method of pesticide application may depend not only on tillage system, but on other factors as well such as the type and properties of field crops planted along with climate and geographic conditions. No-till conservation tillage system is generally the most popular tillage practice in the Maumee River Basin area except for three watershed areas (Findlay, Paulding, and Van Wert) where intensive-till is the most popular tillage system.

Treatment Plant	*Pesticide Use*	*No-till*	*Ridge-till*	*Mulch-till*	*Reduced-till*	*Intensive-till*
Archbold	180	42.92%	0.45%	17.74%	14.94%	23.95%
Bowling Green	4,471	42.62%	0.14%	11.17%	8.91%	37.16%
Defiance	243	46.88%	0.00%	5.49%	9.58%	38.05%
Findlay	170	36.44%	0.13%	13.37%	11.98%	38.08%
Lima	132	41.36%	0.12%	15.15%	10.99%	32.38%
McClure	4,321	42.74%	0.13%	11.14%	8.88%	37.11%
Napoleon	3,978	42.68%	0.11%	10.88%	8.88%	37.45%
Paulding	121	40.22%	0.00%	9.93%	7.07%	42.77%
Swanton	27	46.17%	1.08%	14.93%	8.42%	29.40%
Van Wert	21	42.21%	0.00%	9.40%	3.82%	44.57%
Wauseon	6	46.17%	1.08%	14.93%	8.42%	29.40%
Average	*1,243*	*42.76%*	*0.29%*	*12.20%*	*9.26%*	*35.49%*

Table 1.5: Pesticide Use (in 1,000 pounds) and Tillage Practice
All data are averages of five-year period (1995-1999).

Treatment Plant	*Total Operating Cost*	*Total Chemical Cost*	*Average Operating Cost*[1]	*Average Chemical Cost*[1]	*Pesticide Contamination*[2]
Archbold	$1,173,490	$113,784	$1,741	$166	1.77
Bowling Green	$972,511	$205,043	$601	$127	5.34
Defiance	$3,333,325	$224,763	$2,185	$147	8.41
Findlay	$1,749,190	$191,581	$673	$73	2.28
Lima	$1,930,812	$354,854	$375	$69	3.32
McClure	$158,235	$4,035	$6,294	$160	8.47
Napoleon	$450,893	$74,285	$907	$149	8.98
Paulding	$289,056	$36,204	$1,826	$227	3.16
Swanton	$428,523	$25,993	$2,255	$139	1.23
Van Wert	$945,974	$72,963	$1,474	$114	3.99
Wauseon	$456,003	$43,751	$1,283	$123	3.20
Average	*$1,080,728*	*$122,478*	*$1,783*	*$136*	*4.56*

1. per million gallons of water treated; 2. micrograms per liter in treated water

Table 1.6: Water Treatment Costs and Water Quality
Cost (in year 2000 dollar) and pesticide data are averages of five-year period (1995-1999).

Measures of physical (pesticide contamination levels) and economic (water treatment costs) consequences of agricultural practices on nearby and downstream communities are illustrated in Table 1.6. Total operating cost includes chemical cost, power cost, maintenance cost, pesticide monitoring cost, and other costs. Average operating cost per million gallons of treated water varies more substantially among water treatment plants than average chemical cost; however, it is less negatively correlated with total volume of water treated (correlation coefficient of -0.47) than average chemical cost (correlation coefficient of -0.68). Thus, non-chemical operating costs (power cost, maintenance cost, pesticide monitoring cost, and other costs) must also vary significantly among water treatment plants although this variation may not be directly or significantly related with pesticide use and tillage practice. *Average chemical cost* per million gallons of water treated, including costs for activated carbon, alum, chlorine, and polymers used to clarify and balance the pH of water, is used in the analysis to represent water treatment cost aspect of agricultural externality since it is assumed to be most directly related with the quality of surface water to be treated. Annual average chemical cost per million gallons of water treated in the Maumee River Basin area ranges from $69 to $227 with the average of $136.

As an attempt to measure water quality aspect of agricultural externality, a *pesticide contamination level* (measured as micrograms per liter in finished public surface water) is derived by summing contamination levels of seven commonly applied pesticides; alachlor, atrazine, metolachlor, metribuzin, simazine, cyanazine, and acetochlor. The U.S. EPA has established maximum contaminant levels (MCLs) of some pesticides in drinking water, which are enforceable standards based on a lifetime of exposure. Compliance with the MCL is based on a public water system's running annual average of all samples taken during a 12 month period, and the consumption of water with chemical concentrations less than or equal to MCLs for the duration of time covered by the criteria or standard is considered by the U.S. EPA to pose negligible health risks. Currently, MCLs have been established for only three pesticides, (alachlor (MCL=2), atrazine (MCL=3), and simazine (MCL=4)) out of seven pesticides included in the calculation of the level of pesticide contamination in the finished water of public surface water systems. Pesticide contamination levels vary considerably ranging from 1.23 to 8.98 with the average of 4.56 micrograms per liter. The relationship between average chemical cost per million gallons of water treated and pesticide contamination level in finished public surface water does not show any clearly noticeable pattern with a correlation coefficient of only 0.18.

1.7 Empirical Models

Two empirical models are formulated in an attempt to evaluate relationships between agricultural practices and both surface water quality and water treatment costs in nearby and downstream communities. Pesticide contamination level in finished public surface water and average chemical cost of treating million gallons of surface water (as measures of water quality and water treatment cost effects respectively) are two types of agricultural externality considered in the analysis. Agricultural practices considered include tillage system and pesticide application to cropland. As explained above, conservation tillage may decrease the contamination level of pesticides in surface water by reducing soil erosion and then water runoff due to an increased water infiltration (Baker and Laflen 1983; Fawcett, Christensen, and Tierney 1994). Some water treatment plant characteristics (volume treated and raw water storage capacity) and site-specific environmental factors (precipitation and soil erodibility factor index) are also included in empirical models. Two empirical models estimated are

(1.1) *PESTCNT* = *function* (pesticide use, tillage practice, storage capacity, precipitation, *K* factor)

(1.2) *ACC* = *function* (pesticide use, tillage practice, volume of water treated, precipitation, *K* factor)

where PESTCNT (in micrograms per liter) and ACC (in year 2000 dollar) denote annual average pesticide contamination level in finished surface water and average chemical cost of treating million gallons of water respectively. Non-agricultural pesticide uses, such as pesticide application to home lawns and golf courses, and other topographical characteristics that may influence pesticide contamination levels and water treatment costs are not included in both models due to data limitations. Omission of relevant pesticide use and environmental characteristics variables could result in biased coefficient estimates of included variables, and the direction of the bias depends on the correlation between the omitted variable and all included variables as well as on the sign of the true coefficient of the omitted variable.

1.8 Model Estimation

Since the data include 11 water treatment plants and corresponding watershed areas for the five-year sample period, possibilities of plant or site-specific and year-specific effects should be examined first before deciding whether we are able to pool the data or not. In our empirical models, equations (1.1) and (1.2), *K* factor and raw water storage capacity represent time-invariant, watershed area and water treatment plant-specific variables respectively. These two variables are included to capture the fixed effect of specific watershed area and water treatment plant on pesticide contamination level and/or average chemical cost of treating surface water. To examine the possibility of year-specific economy wide effect, a full model including dummy variables for each sample year is estimated to perform an F-test of H_0*: all year dummies are zero*. The result of this test implies that there is no statistically significant year-specific effect; therefore, cross sectional data for 11 water treatment plants and corresponding watershed areas are pooled together across five sample years with some variables to capture water treatment plant and watershed area-specific effects. All pooled models estimated include time-invariant variables, *K* factor and/or raw water storage capacity, to account for site and/or plant-specific effects.

First, two versions of *water quality effect model* are estimated with different classifications of tillage practices, one with all five detailed tillage systems and the other with two broad categories of conservation and conventional tillage systems, to investigate the relationship between agricultural practices and finished surface water quality. Conservation tillage includes no-till, ridge-till, and mulch-till systems and conventional tillage includes reduced-till and intensive-till systems. Two estimated water quality models in terms of pesticide contamination level are

(1.3) $$PESTCNT = \beta_0 + \beta_1 PEST\_TH + \beta_2 PEST\_TH^2 + \beta_3 CSN\_TH + \beta_4 CSR\_TH + \beta_5 CSM\_TH + \beta_6 CV30\_TH + \beta_7 CV15\_TH + \beta_8 RSTOR\_ML + \beta_9 PREC + \beta_{10} KFACT$$

(1.4) $PESTCNT = \beta_0 + \beta_1 PEST\_TH + \beta_2 PEST\_TH^2 + \beta_3 CS\_TH + \beta_4 CV\_TH + \beta_5 RSTOR\_ML + \beta_6 PREC + \beta_7 KFACT$

where PEST_TH is pesticide use in 1,000 pounds, CSN_TH is conservation no-till in 1,000 acres, CSR_TH is conservation ridge-till in 1,000 acres, CSM_TH is conservation mulch-till in 1,000 acres, CV30_TH is reduced-till in 1,000 acres, CV15_TH is intensive-till in 1,000 acres, CS_TH is conservation tillage in 1,000 acres, CV_TH is conventional tillage in 1,000 acres, RSTOR_ML is raw water storage capacity in million gallons, PREC is total annual precipitation in inches, and KFACT is soil erodibility factor index.

Next, two similar versions of *water treatment cost model* are also estimated to investigate the relationship between agricultural practices and water treatment costs. Two estimated water treatment cost models in terms of average chemical cost per million gallons of water treated are

(1.5) $ACC = \beta_0 + \beta_1 PEST\_TH + \beta_2 PEST\_TH^2 + \beta_3 CSN\_TH + \beta_4 CSR\_TH + \beta_5 CSM\_TH + \beta_6 CV30\_TH + \beta_7 CV15\_TH + \beta_8 VOL\_ML + \beta_9 VOL\_ML^2 + \beta_{10} PREC + \beta_{11} KFACT$

(1.6) $ACC = \beta_0 + \beta_1 PEST\_TH + \beta_2 PEST\_TH^2 + \beta_3 CS\_TH + \beta_4 CV\_TH + \beta_5 VOL\_ML + \beta_6 VOL\_ML^2 + \beta_7 PREC + \beta_8 KFACT$

where VOL_ML is a volume of water treated in million gallons. Both water quality and water treatment cost models are estimated using the Maumee River Basin data pooled across five years of sample period (1995-1999) along with time-invariant, watershed area (*KFACT*) and water treatment plant (*RSTOR_ML*)-specific variables.

1.9 Estimation Results

The results of pooled ordinary least squares estimation are summarized in Tables 1.7 and 1.8 for water quality and water treatment cost models respectively with two different classifications of tillage practices. In the Maumee River Basin area, conservation no-till system has been adopted on more than 40 percent of total planted acres, while conservation ridge-till and conservation mulch-till systems have been adopted on less than 1 percent and about 11 percent of total planted acres respectively throughout whole sample period. Therefore, the effect of CS_TH (the sum of all three conservation tillage systems) on surface water quality and water treatment costs could be dominated by the effect of conservation no-till due to such a dominant adoption rate. The effect of intensive-till system, which has been adopted on more than 35 percent of total planted acres, could also dominate the effect of CV_TH (the sum of two conventional tillage systems). To examine whether an independent effect of each tillage system exists or not, all five tillage systems are separately estimated first, and three conservation tillage (CS_TH) and two conventional (CV_TH) tillage systems together are then estimated in both water quality and water treatment cost models.

1.9.1 Water Quality Model

Table 1.7 shows the estimation results from two different versions of the water quality model using average annual pesticide contamination level in finished surface water as a measure of water quality. Pesticide application to crop field has a positive effect on pesticide contamination level in surface water although significance level is not high in both equations (1.3) and (1.4). No-till conservation tillage is expected to reduce pesticide contamination level in surface water by reducing soil erosion and, in turn, runoff water due to crop residue covered on the soil surface. As expected, conservation no-till (CSN_TH) in equation (1.3) has a significant, negative effect on pesticide contamination level, implying that an increase in conservation no-till system by 1,000 acres in a typical watershed area will reduce pesticide contamination level by 0.046 micrograms per liter which is about 1 percent of the average level of pesticide contamination (4.56 micrograms per liter) in the Maumee River Basin. On the contrary, both types of conventional tillage systems, reduced and intensive tillage systems, are shown to increase pesticide contamination level by a similar magnitude with a no-till's negative effect on pesticide contamination level. An increase in reduced-till and intensive-till systems by 1,000 acres in a typical watershed area will lead to about 1.1 percent and 1 percent increases in the average annual level of pesticide contamination respectively.

In equation (1.4), effects of both conservation tillage and conventional tillage on the quality of surface water measured by pesticide contamination level are significant and signs are as expected. Three conservation tillage systems collectively have a negative effect on pesticide contamination level at a similar magnitude with a no-till's effect alone, about 1 percent reduction in the average annual level of pesticide contamination with an increase of any conservation tillage by 1,000 acres. Water quality effect of 1,000 acres increase in conservation tillage is predicted to range from 0.5 percent (Napoleon watershed area) up to 3.8 percent (Swanton watershed area) reduction in the average annual level of pesticide contamination. Two conventional tillage systems collectively have a positive effect on pesticide contamination level at a magnitude slightly higher than individual effects of reduced-till and intensive-till systems, about 1.2 percent increase in the average annual level of pesticide contamination with an increase of any conventional tillage by 1,000 acres. Water quality effect of 1,000 acres increase in conventional tillage is predicted to range from 0.6 percent (Napoleon watershed area) up to 4.3 percent (Swanton watershed area) rise in the average annual level of pesticide contamination.

Raw water storage capacity of water treatment plant is expected to decrease pesticide contamination level in surface water because storage reservoirs allow dissolved pesticides contained in runoff water to dilute and settle in the water. In both equations (1.3) and (1.4), raw water storage capacity shows a significant, negative effect on pesticide contamination level although the magnitude of the effect is not large.

Climate and geological factors included to capture site-specific effects on surface water contamination show relatively significant effects in both equations (1.3) and (1.4). Precipitation is expected to increase the level of pesticide contamination in surface water by increasing the volume of runoff water holding all other factors constant. It may also be expected to decrease pesticide concentration level by increasing the volume of raw water in storage reservoir. The effect of increased runoff water (increased pesticide concentration) appears to be more dominant than the effect of increased raw water in storage reservoir (decreased pesticide concentration) showing a relatively significant, positive effect of precipitation. Soil erodibility factor, *K*, is expected to increase pesticide concentration level by increasing the volume of runoff water since a higher *K* implies that soil particles are more susceptible to detachment and movement by water. The sign of the estimated coefficient of soil erodibility factor is as expected although the level of statistical significance is not as high as precipitation.

Variables	*Coeff.*	*t value*	*Pr > |t|*	*Variables*	*Coeff.*	*t value*	*Pr > |t|*
Intercept	-9.8371	-1.85	0.0707	**Intercept**	-9.4287	-1.86	0.0687
PEST_TH	0.0098	1.24	0.2211	**PEST_TH**	0.0079	1.39	0.1699
PEST_TH²	-0.000002	-2.07	0.0445	***PEST_TH²***	-0.000001	-2.26	0.0285
CSN_TH	-0.0460	-3.27	0.0021	***CS_TH***	-0.0474	-5.36	<.0001
CSR_TH	0.2019	0.1	0.9189				
CSM_TH	-0.0393	-0.6	0.5514				
CV30_TH	0.0478	2.96	0.0049	***CV_TH***	0.0524	5.34	<.0001
CV15_TH	0.0435	1.96	0.0562				
RSTOR_ML	-0.0004	-2.37	0.0223	***RSTOR_ML***	-0.0004	-2.39	0.0209
PREC	0.1322	1.66	0.1032	***PREC***	0.1350	1.76	0.0849
KFACT	27.2989	1.55	0.1272	**KFACT**	25.9018	1.55	0.1283

Table 1.7: Water Quality Models: Equations (1.3) and (1.4)
PESTCNT (pesticide contamination level) is a dependent variable.
R^2 = 0.6481 and 0.6453 for equations (1.3) and (1.4)

1.9.2 Water Treatment Cost Model

Table 1.8 shows the estimation results from two different versions of the water treatment cost model using annual chemical cost per million gallons of water treated as a measure of water treatment cost which is assumed to be most directly related to the pesticides dissolved in runoff water. Compared to water quality models, the amount of pesticide application to cropland shows a statistically more significant, positive effect on average chemical cost especially in equation (1.5), while all tillage systems, either independently or collectively, do not show any statistically significant effect in water treatment cost models. According to equation (1.5), a decrease in pesticide use by 1,000 pounds in a typical watershed area will lead to a reduction of $.21 in annual average chemical cost per million gallons, implying an annual saving of $261 with 1,226 million gallons treated per year on average. Water treatment cost effect of 1,000 pound decrease in pesticide application is predicted to range from $6 (McClure watershed area) up to $1,102 (Lima watershed area) reduction in annual total chemical cost of water treatment.

In both equations (1.5) and (1.6), average chemical cost is shown to decrease with a volume of water treated and this negative effect becomes smaller as total volume of water treated increases, implying a conventional U-shaped average cost curve with respect to quantity. The convexity of average chemical cost curve is well supported by the data with negative and positive signs associated with VOL_ML and VOL_ML^2 respectively.

In equation (1.5), geological factor (soil erodibility index) shows a significant, positive effect as expected while climate factor (precipitation) does not show any statistically significant effect on average chemical cost in both equations (1.5) and (1.6).

Soil erodibility factor seems to have a more significant effect than precipitation on average chemical cost, while precipitation shows a more significant effect than soil erodibility factor on pesticide contamination level.

Variable	*Coeff.*	*t value*	*Pr > |t|*	*Variable*	*Coeff.*	*t value*	*Pr > |t|*
Intercept	40.4214	0.57	0.5711	Intercept	70.6611	1.04	0.3021
PEST_TH	0.2132	2.04	0.0476	PEST_TH	0.1079	1.41	0.1661
PEST_TH²	-0.00002	-1.96	0.0565	PEST_TH²	-0.00001	-1.38	0.1742
CSN_TH	0.0710	0.38	0.7073	CS_TH	-0.1231	-1.03	0.3072
CSR_TH	39.78	1.5	0.1399				
CSM_TH	-1.12	-1.28	0.2062				
CV30_TH	-0.1637	-0.77	0.4433	CV_TH	-0.0580	-0.44	0.663
CV15_TH	-0.4014	-1.36	0.1815				
VOL_ML	-0.0500	-4.48	<.0001	*VOL_ML*	-0.0466	-4.3	<.0001
VOL_ML²	0.000005	2.37	0.0225	*VOL_ML²*	0.000004	2.18	0.0348
PREC	0.0593	0.06	0.9552	PREC	0.2105	0.2	0.8401
KFACT	405.6707	1.75	0.0872	KFACT	299.7148	1.36	0.1801

Table 1.8: Water Treatment Cost Models: Equations (1.5) and (1.6)
ACC (average chemical cost per million gallons of water) is a dependent variable.
R^2 = 0.5791 and 0.5552 for equations (1.5) and (1.6)

1.10 Conclusions

Surface water quality and water treatment cost aspects of "agricultural externality" caused by farming practices are evaluated using pesticide contamination level in finished public surface water and average chemical cost per million gallons of water treated in the Maumee River Basin located in northwestern Ohio, northeastern Indiana, and southern Michigan. The results clearly indicate that there are statistically significant relationships between farming practices, such as pesticide use and tillage practice, and both surface water quality and water treatment costs.

Several studies attempted to identify the relationship between the costs of treating surface water and water quality based on either turbidity or sediment loading although they used somewhat different types of water treatment costs. Dearmont, McCarl, and Tolman (1998) estimate the chemical costs of $74.15 per million gallons of water treated along with estimated cost elasticity for water turbidity of 0.27. Forster, Bardos, and Southgate (1987) estimate the variable costs of $92.28 per million gallons of water treated with cost elasticities for turbidity and soil erosion rate of 0.119 and 0.406 respectively. Holms (1988) estimates total costs of $113.12 per million gallons of water treated, including operating and maintenance costs, with cost elasticity for turbidity of 0.07. Moor and McCarl (1987) estimate the costs for alum, lime, and sediment removal of $20 per million gallons of water treated with cost elasticity for turbidity of 0.333.

Compared to these existing studies on the relationship between water treatment costs and turbidity, results from the Maumee River Basin area provide estimated average chemical costs of $136 per million gallons of water treated, implying somewhat higher water treatment costs in this area than suggested by other studies. Instead of estimating cost elasticity for turbidity as in other studies, this study allows us to uniquely estimate water treatment cost elasticity for pesticide use and water quality elasticity for conservation tillage adoption. Findings are that average chemical cost per million gallons decreases by 1.95 percent for a 1 percent reduction in pesticide application to cropland (equation (1.5)), while pesticide contamination level in finished surface water decreases by 4.32 percent for a 1 percent more adoption of any conservation tillage system (equation (1.4)) in a typical watershed area of the Maumee River Basin based on sample mean values and coefficient estimates. If the relationship between communities' health-related costs and pesticide concentrations in surface water could be identified, these results would be more meaningful for policy analysis by allowing us to relate agricultural practices to both water treatment and health-related costs.

Water quality and water treatment cost benefits from more adoption of conservation tillage and less use of pesticides are clearly demonstrated in terms of a reduction in pesticide contamination level in finished surface water and a decrease in average chemical cost of treating surface water. However, it is possible that there is a conflict between more conservation tillage and less pesticide use since conservation tillage system may have to rely more on pesticides for weed, insect, and fungal controls.

More detailed and accurate data for climate, geological, and geographic characteristics could be incorporated into the empirical models estimated here if these data can be made available. For example, more detailed soil properties, seasonal patterns of precipitation events, and potential interactive effects of these environmental factors with farming practice variables can be used to explain the variation in surface water quality and water treatment costs. The timing of pesticide application coupled with seasonal patterns of precipitation will significantly influence the amount of pesticide runoff. Especially, heavy rainfall events right after pesticide application to cropland will seriously increase pesticide runoff into surface water, ultimately affecting both pesticide contamination level in surface water and water treatment costs. According to the U.S. Geological Survey (USGS), pesticide concentrations in surface waters follow strong seasonal patterns in agricultural areas due to the timing of pesticide application and runoff conditions.

One potentially important aspect of agricultural externality not addressed in this study is pesticide contamination in ground water that is used for drinking water source by about 50 percent of the Nation's population. Pesticides contaminated in ground water mainly through the water infiltration into the soil surface could impose risks to human health, aquatic ecosystems, and wildlife, let alone a possible increase in water treatment costs. Possible effects of pesticide contamination in both surface and ground waters on wildlife and aquatic ecosystems could be further explored as another set of externality costs associated with agricultural practices.

ESSAY 2

THE ECONOMIC VALUE OF MARINE RECREATIONAL FISHING: APPLYING BENEFIT TRANSFER TO MARINE RECREATIONAL FISHERIES STATISTICS SURVEY (MRFSS)

ABSTRACT

A comprehensive survey of current literature on benefit transfer is conducted as an attempt to answer a question of when, why, and how to use this technique. Then, benefit function transfer technique is applied to the estimation of two welfare measures (access value to fishing sites and willingness to pay for better fishing experience) in a marine recreational fishing environment of the coastal states in the Northeast and Southeast U.S. Using two data sets from the same source but in different years (1994 and 1997) and regions (Northeast and Southeast), benefit transfer estimates are compared with original estimates to examine the convergent validity of benefit function transfer. Although benefit transfer error could go up to over 400% of original estimates for some cases, the magnitude of benefit transfer error is less than 100% of original estimates for most cases. Since two data sets used for benefit transfer are from different regions and years, whether regional or temporal variation is more responsible for benefit transfer error can not be determined with current data.

2.1 Introduction

Benefit transfer generally refers to the practice of applying estimates of economic value obtained from one or more original valuation studies in one context to the evaluation of economic value in another context by *adaptively transferring* available information (value estimates or estimated benefit/demand function) from existing primary studies. Following Desvousges, Naughton, and Parsons (1992), a place for which original research was conducted is called a "*study site*" and a place to which estimates of economic value from original research are transferred is called a "*policy site*." As a less costly and time saving method of obtaining estimates of non-market value for various outdoor recreation activities, the primary goal of benefit transfer practice is to estimate economic benefits of non-market activities with an acceptable degree of accuracy for one context (a policy site) by adaptively transferring benefit estimates or a benefit function from some other context (a study site) when it is too costly or takes too much time to conduct a primary valuation study.

Benefit transfer provides a means by which economic value of an outdoor recreation activity at an unstudied policy site can be estimated using information available from a study site(s). For instance, economic value of marine recreational fishing in a particular state or region could be estimated by transferring estimates of economic value of marine recreational fishing from the original valuation study conducted in another state or region after adjusting to new circumstances (policy site context), especially to different characteristics of angler population and fishing sites. Although this study focuses on transferring economic estimates of non-market value of marine recreational fishing, benefit transfer techniques discussed here could be more broadly applied in a number of other outdoor recreation activities. By providing preliminary measures of economic value estimates in various circumstances, benefit transfer may also be applied in screening agricultural policies, evaluating environmental policies (e.g., U.S. Environmental Protection Agency's (EPA) (1997) assessment of the Clean Air Act), defining the extent of the market affected by a proposed policy, initial screening of natural resource damage assessment, and determining whether original research is warranted (Rosenberger and Loomis 2003).

After a comprehensive survey of the current literature on benefit transfer, benefit transfer technique is applied to the estimation of marine recreational fishing value in the Northeast and Southeast coastal regions of the United States using data from the National Marine Fisheries Service's (NMFS) Marine Recreational Fisheries Statistics Survey (MRFSS) combined with the Add-On MRFSS Economic Survey (AMES) in 1994 and 1997 respectively. The *convergent validity* of benefit transfer is examined by comparing the value estimates obtained from benefit transfer procedures to the value estimates obtained from original non-market valuation research.

2.2 Historical Background of Benefit Transfer

Even before any attempt to develop formal terminology or systematic procedures and protocols and definitely before any rigorous testing of the validity and reliability of the methodology, the practice of benefit transfer became popular in the economic analysis of the consequences of environmental regulations in the United States during the mid

1980s. The U.S. Army Corps of Engineers, the U.S. Bureau of Reclamation, and the U.S. Forest Service identified a need for estimates of economic value associated with various outdoor recreation activities for the purpose of formal project evaluation and planning.

The U.S. Water Resources Council first published unit day value estimates for various recreation activities to evaluate water-related projects in 1973, and updated these recreational value estimates in 1979 and 1983. The U.S. Forest Service also began publishing Resources Planning Act (RPA) values for recreation in 1980 as per person per activity day estimates driven by the Renewable Resources Planning Act of 1974 which requires an assessment of the supply of and demand for renewable resources on the Nation's forests and rangelands along with a formal analysis of the costs and benefits associated with the United States Department of Agriculture (USDA) Forest Service's programs. These requirements of an assessment of renewable resources and a cost benefit analysis of programs create the need for accurate and reliable measures of non-market value of various outdoor recreation activities. Both the U.S. Water Resources Council's unit day recreation values and the U.S. Forest Service's RPA recreation values are based primarily on average willingness to pay estimates for various outdoor recreation activities derived from past empirical studies along with expert judgment and political screening.

A major development in benefit transfer occurred in 1992 with the publication of a special section on benefit transfer in the journal *Water Resources Research*. This special section collectively provides an extensive review and critique on benefit transfer methods by defining theories, identifying needs, suggesting protocols, and presenting new approaches. Brookshire and Neill (1992) provide an introduction and overview of this special section on benefit transfer, focusing on conceptual and empirical issues. They address some fundamental issues regarding benefit transfer, including limitations and the need for protocol development. They also point out that the most critical limitation of benefit transfer practice: *benefit transfer estimates can only be as accurate as the original estimates of economic benefits*. Existing problems associated with original non-market valuation studies will be only magnified in the application of benefit transfer. One seemingly common conclusion from the papers in this special issue seems that benefit transfer is valid only under well-defined conditions although authors don't appear to argue about the possibility of the practice. Prior to this special section, most benefit transfer applications used a *value transfer* method that either directly or adaptively (e.g., day use values adjusted for population or site characteristics) transfers a point estimate(s) or a central tendency measure of original study estimates. Loomis (1992) proposes a *benefit function transfer* method that transfers an entire demand, benefit, or willingness-to-pay function as a more rigorous and robust method of benefit transfer compared with the simple transfer of a point estimate(s) or a measure of central tendency.

A number of formal studies have been investigating the application of non-market valuation methods (e.g., travel cost method, hedonic pricing method, and contingent valuation method) and the validity of various benefit transfer approaches (e.g., value transfer, function transfer, and meta-analysis function transfer) since the publication of a special section on benefit transfer in 1992. Many studies empirically try to evaluate the validity and reliability of benefit transfer applications in various contexts by either adopting a benefit function transfer method or comparing it with a value transfer method (Parson and Kealy 1994; Loomis et al. 1995; Downing and Ozuna 1996; Feather and Hellerstein 1997; Kirchhoff, Colby, and LaFrance 1997; Brouwer and Spaninks 1999; Piper and Martin 2001; Smith, Houtven, and Pattanayak 2002). The application of a benefit function transfer method always seems to perform better than a simple value transfer method, providing empirically more valid and reliable benefit transfer estimates for the policy site. O'Doherty (1995), Desvousges, Johnson, and Banzhaf (1998), Bergstrom and De Civita (1999), Brouwer (2000), and Rosenberger and Loomis (2001, 2003) provide comprehensive overviews of the current status of benefit transfer as a potentially cost and time efficient non-market valuation technique.

2.3 An Overview of Benefit Transfer Methodology

Benefit transfer is a practical methodology in evaluating the economic consequences of environmental policies and programs with an underlying assumption that economic benefits and/or costs associated with a particular environmental commodity or change could be extrapolated from existing valuation studies of similar context. In possibly many circumstances, primary research may not be justified or plausible due to budget constraint and/or time limitation necessitating the application of an alternative benefit transfer method. However, this low cost and less time-consuming alternative method for non-market valuation may only be valid and reliable under special circumstances. In addition, there are also several important limitations associated with the application of benefit transfer even when these special circumstances are satisfied. Before a thorough discussion of general procedures involved with performing and checking the validity of benefit transfer, we need to carefully examine the circumstances under which benefit transfer methods can be meaningfully carried out and potential advantages and limitations of these methods.

2.3.1 Necessary Conditions for Successful Benefit Transfer

In order to carry out economically meaningful benefit transfer when primary research for a policy site is not a plausible option, there are some necessary conditions that should be satisfied (Desvousges, Naughton, and Parsons 1992; Rosenberger and Loomis 2001).

First, the policy site (the region to which estimated economic benefits are transferred) context should be thoroughly defined. The extent, magnitude, and quantification of the expected impacts from the proposed policy action on the policy site or on its resources should be identified along with the extent and magnitude of the population that will be affected by the expected changes in the characteristics of the policy site. The availability of current primary and/or secondary data at the policy site and further data needs for benefit transfer application should be identified, including the type of measurement (unit, average, or marginal value), the kind of value measured (use, nonuse, or total value), and the degree of certainty surrounding the transferred information (i.e., the accuracy (the closeness of a measurement to the true or accepted value) and precision (the reproducibility of multiple measurements described by the standard error or confidence interval) of transferred estimates).

Second, the study site (the region from which estimated economic benefits are transferred) should meet certain conditions for successful benefit transfer. It is necessary that original studies transferred should be based on adequate data, sound economic method, and correct empirical technique (Freeman 1984). The statistical relationships between economic benefits (or costs) and both socio-economic characteristics of the affected population and physical/environmental characteristics of the study site should be contained in the original valuation study. In addition, an adequate number of original studies on a particular recreation activity for similar sites would allow us to carry out more reliable statistical inferences regarding the transferability of estimated values from the study site(s) to the policy site.

Finally, the study site(s) and the policy site should exhibit an adequate level of similarity in terms of the environmental resource evaluated, the nature of an environmental change, and the characteristics of the affected populations and sites. The conditions and quality of the recreation activity analyzed should be similar, including intensity, duration, and skill requirement. Unless enough information on own and substitute prices is available, the markets for the study site and the policy site should be similar. The quality and quantity of the change in the environmental resource at the study site should be similar to those of the expected change in the environmental resource at the policy site, including the measurability and the source of the change. Other important characteristics that should be considered include site and population characteristics. The similarity of socio-economic profiles of the affected populations and the characteristics of the environmental resource of interest between the study site and the policy site is an important requirement for a successful application of benefit transfer. Benefit transfer applications work more effectively and efficiently when the attributes of the environmental resource, the nature of the environmental change, the characteristics of the affected populations and sites display an adequate level of similarities between the study site and the policy site.

The above information requirements to implement effective and efficient benefit transfer applications are not always satisfied in the reporting of data and estimation results from primary studies since most original research is not conducted for the future purpose of transferring estimated economic benefits or costs to the policy site of similar context. Therefore, the implicit costs of performing benefit transfer with incomplete information should be deliberately accounted for as well as the potential benefits of additional information from possibly expensive primary research.

2.3.2 Potential Advantages of Benefit Transfer

Benefit transfer is a typically inexpensive and quicker method of obtaining economic benefit estimates of various recreation activities compared with an original non-market valuation study as mentioned above. A successful application of benefit transfer could have some other advantages in addition to the obvious resource and time advantages (Desvousges, Johnson, and Banzhaf 1998).

First, the benefit transfer method systematically incorporates economic benefits into benefit-cost analysis by calculating costs and/or benefits in a way that is consistent with economic theory by recognizing the behavioral relationships of non-market recreation activities with recreator's socio-economic variables and recreation site's physical characteristics. Transferred benefits for the policy site are constructed based primarily on the benefit estimates derived from non-market valuation techniques that systematically explain various recreation activities as a function of population characteristics and recreation site's physical attributes.

Second, the benefit transfer methodology can help us organize policy issues by providing a logical framework for non-market valuation while remaining flexible. As new policy issues are identified or new original valuation studies become available, benefit transfer methods can readily allow additional branches to be attached to the existing framework by adapting the calculations of benefits to these new circumstances. In the practice of benefit transfer methods, new aspects of recreation benefits or environmental costs and better original studies as they become available could be easily incorporated into the existing structure of benefit transfer. The benefit transfer method identifies and uses the appropriate variables for measuring these new aspects of benefits or costs with existing or improved, as better original studies become available, framework.

Finally, the benefit transfer method could be used as a screening technique to determine whether a more detailed, primary valuation study should be conducted. The availability of relatively inexpensive and quick estimates of various recreational values through the application of benefit transfer methods allows researchers or resource managers to focus their efforts more efficiently and effectively. As the accuracy and precision of transferred benefit estimates for the policy site increase, the need for expensive and time-consuming primary research will necessarily decrease.

2.3.3 Potential Limitations of Benefit Transfer

Several studies (Boyle and Bergstrom 1992; Desvousges, Naughton, and Parsons 1992; Navrud and Pruckner 1997; Desvousges, Johnson, and Banzhaf 1998; Bergstrom and De Civita 1999; Azqueta and Touza 2000; Brouwer 2000; Rosenberger and Loomis 2001) collectively provide a comprehensive overview on potential problems associated with the application of benefit transfer methods.

First, the most fundamental limitation of benefit transfer methods stems from the quality of the original valuation study. Brookshire and Neill (1992) point out that benefit transfer estimates cannot be more reliable than the original study estimates upon which they are based, and the problems associated with the original non-market valuation study will only be magnified in the benefit transfer process. As pointed out by many researchers, the quality of the original research significantly affects the quality of benefit transfer procedure: “garbage in, garbage out” factor (Rosenberger and Loomis 2001). Because most primary valuation studies are not designed for the future application in the benefit transfer process, the inadequacy of the reporting of original valuation studies could also influence the quality of the benefit transfer process by preventing us from adapting estimated non-market values to possibly different circumstances of the policy site. Although there are no clear guidelines for evaluating the quality of original studies, both Desvousges, Naughton, and Parsons (1992) and Boyle and Bergstrom (1992) suggest some criteria that help researchers select among available original valuation studies for benefit transfer. Their criteria include adequate data, sound economic method, and correct empirical technique; similarities between the study site and the policy site in terms of non-market activity, environmental change, and relevant markets and populations affected; description of non-market value as a function of socio-economic variables and site characteristics; and proper assignment of property rights leading to the same theoretically appropriate welfare measures at both study and policy sites.

Second, an important limitation can also arise from the availability of relevant original valuation studies. Finding appropriate valuation studies that correspond to the policy site context, especially with regard to site characteristics and available substitutes, could be difficult because many original valuation studies are single-site studies with no substitutes and no variation in site characteristics, or available multi-site studies may not include the site similar to the policy site. For some recreation activities, only a small number of original valuation studies may exist although this issue can be improved through time as more original non-market valuation studies based on primary data are implemented by providing a greater pool of non-market value estimates upon which benefit transfer could be based. As more original valuation studies are conducted, these studies could be made more easily accessible to the researchers conducting benefit transfer studies by establishing a nationwide or worldwide database system of both published and unpublished non-market valuation studies containing data sets, estimation techniques, and actual welfare estimates.

Third, the degree of correspondence between the study site and the policy site can affect the efficiency and effectiveness of benefit transfer methods. Benefit transfer could produce inaccurate benefit estimates due to the lack of similarities between the study site and the policy site in terms of site and population-specific characteristics: site quality, the degree of quality change, site location, and socio-economic characteristics of affected populations. Some original studies may estimate different non-market values of particular recreation activities at unique recreation sites under unique circumstances, leading to quite different estimated values. Different temporal and spatial dimensions of the study and policy sites, let alone among original studies, could affect the stability of data and value estimates over time and across locations. Since existing valuation studies usually occur at different points in time and/or locations, the extent of the affected populations and resources may not be directly comparable, necessitating the need for adjustments for meaningful benefit transfer.

Fourth, many subjective judgments, sometimes inevitably, involved in the process of benefit transfer may affect the validity and/or reliability of value estimates obtained from benefit transfer. Usually, benefit transfer practitioners should make a number of assumptions and professional judgments in applying benefit transfer methods: "*There is no simple, acceptable way mechanically to transfer a model. Just as the chief ingredient in model construction is judgment, it is also the most important ingredient in transferring* *benefits*" (McConnell 1992). For instance, researchers may often need to make assumptions about how to measure environmental quality and how the proposed changes in measured environmental quality affect behavior. In addition, the crucial assumption for empirically testing the validity of benefit transfer estimates is that the original study estimates available at the policy site are the "true value" of the environmental resource being evaluated, and benefit transfer estimates can be validated by comparing them with the assumed true value (convergent validity test).

Fifth, several methodological issues should be addressed as possible limitations of benefit transfer. Different research and statistical methods used across existing valuation studies could lead to significant differences in estimated values. In estimating non-market value of various recreation activities, original studies may apply revealed (stated) preference techniques which indirectly (directly) estimate consumer surplus (willingness to pay). Revealed preference techniques rely on the *weak complementarity* (no non-use value) assumption between a recreation activity and market goods necessary to participate in the activity, implying that environmental amenity has no effect on the individual's welfare unless market goods required for recreation experience are purchased. Stated preference techniques rely on the constructed *hypothetical markets* through which people's willingness to pay for environmental resources or recreation opportunities are derived. The most popular revealed preference technique is the travel cost method which uses the variable costs of a recreation activity (e.g., transportation, lodging, entrance fees, equipment rentals, and travel time) as a proxy for the price of a non-market recreation activity to derive a travel demand function. The contingent valuation method, which is the most popular stated preference technique, directly asks people their maximum willingness to pay or minimum compensation required for a recreation opportunity or changes in a recreation experience in a hypothetical market. Original studies may estimate different types of non-market value (use value and/or non-use value) using different methodologies (travel cost method and/or contingent valuation method) with different definitions of a relevant market (the size of affected population and the availability of substitutes), making the comparison of various existing studies more difficult and problematic.

Finally, Bergstrom and De Civita (1999) illustrate potential sources of measurement error in value estimates generated by benefit transfer methods: commodity, population characteristics, welfare change, physio-economic linkage, and estimation procedure and judgment. Failure to identify and measure relevant environmental commodities with available substitutes and complements at the policy and study sites could lead to inaccurate benefit transfer estimates. Errors associated with identifying and measuring socio-economic characteristics (e.g., age, education, income, religion, cultural aspects, and family status) of the study and policy sites could introduce measurement error if the characteristics of the affected populations at the study and policy sites are different. The theoretical inconsistency of welfare change measures across the study site and the policy site could cause measurement error in transferring value estimates from the study site. Differences in the relationships between the physical world and the economic behavior and value across the study site and the policy site, coupled with errors in identifying and measuring these linkages, could result in large measurement error.

Errors associated with statistical estimation procedures and subjective professional judgments in adaptively transferring study site values could also lead to benefit transfer measurement error. Any measurement error inherent in the value estimates from the study site will certainly end up being transferred to the policy site as a result of benefit transfer, contributing to measurement errors of benefit transfer estimates mentioned above.

The potential limitations illustrated above could lead to biased benefit transfer estimates and decrease the robustness of benefit transfer procedure. Although original study estimates are approximations themselves and therefore subject to many sources of errors, potential limitations of benefit transfer process itself should be minimized by attempting to identify and control most relevant limitations for each benefit transfer application.

2.3.4 Validity and Reliability of Benefit Transfer

Since benefit transfer methodologies are based on the adaptive use of value estimates from existing non-market valuation studies, the validity and reliability of benefit transfer estimates transferred from the study site to the policy site are fundamental elements of the credibility and success of benefit transfer practices. Although theoretical and empirical aspects of the validity and reliability of transferred value estimates could be evaluated together, the main focus will be on empirical aspect of validity and reliability issues regarding benefit transfer methodologies.

The concept of validity of using value estimates obtained from benefit transfer process as an alternative to conducting original valuation research for the policy site is related with the absence of *biases* in value estimates (Azqueta and Touza 2000). Validity examines the similarity between what needs to be valued (economic value at the policy site) and what is actually valued (transferred economic value from the study site's value estimates). Since it is impossible to directly examine the equality between the "true value" and benefit transfer estimates at the policy site to test the validity of benefit transfer methods, we can indirectly assess validity by using some reference measures (assumed true value) with which benefit transfer estimates can be compared.

Azqueta and Touza (2000) introduce three types of validity concepts depending on the type of reference measures used: criterion validity, theoretical validity, and convergent validity. *Criterion validity* tests use some particular reference measures that may be considered as valid criteria to compare with estimated values. For instance, hypothetical contingent valuation estimates can be compared with outcomes of simulated or actual markets for the same good. *Theoretical validity* focuses on the theoretical determinants of value estimates from various non-market valuation techniques (e.g., travel cost, hedonic pricing, and contingent valuation methods) by examining the consistency of estimation results with underlying economic theories. For example, the magnitude and sign of the estimated coefficients are investigated to check whether they are consistent with economic theories.

Convergent validity, the most popular validity concept in evaluating the value estimates from benefit transfer methods, is based on the comparison of benefit transfer estimates with the value estimates obtained from original valuation study conducted at the policy site. Convergent validity tests generally begin with the evaluation of the degree of statistical equality of estimated coefficients at primary research and transferred benefit functions, both for the policy site. Implicit assumption of coefficient equivalence test is that if the estimated coefficients of two benefit functions are statistically equivalent, benefit estimates derived from these functions are also statistically equivalent. When nonlinearities are present in the willingness to pay or demand functions, these nonlinearities could, however, introduce the case where statistically equivalent benefit functions yield statistically different welfare measures as pointed out by Downing and Ozuna (1996). Because nonlinearities in benefit functions may cause a divergence between statistically equal benefit functions and their respective benefit estimates, convergent validity tests usually involve a second step of examining statistical similarities between benefit transfer estimates and primary research estimates. Convergent validity tests performed in the existing studies (Loomis 1992; Parsons and Kealy 1994; Loomis et al. 1995; Downing and Ozuna 1995; Kirchhoff, Colby, and LaFrance 1997; Brouwer and Spaninks 1999; Rosenberger and Loomis 2001) suggest that benefit function transfer is more robust than value transfer, implying the importance of systematic adjustment of study site values to the differences in the characteristics of affected populations and sites between the study site and the policy site.

The reliability of non-market value estimates derived from various valuation techniques, including benefit transfer method, is related with the *variance* of estimated monetary values (Azqueta and Touza 2000). When the variance of estimated non-market value is large, estimated benefit measures are considered as unreliable. The reliability of benefit estimates does not guarantee the validity of these estimates since reliability is a necessary but not sufficient condition for validity which requires unbiasedness of value estimates. Loomis (1989) points out that reliable (small variance) value estimates could be biased upward or downward; however, reliability may indicate that the reported value estimates are consistent, and reflect a substantial deterministic component of respondent's behavior.

O'Doherty (1995) indicates that we can interpret some early tests of the reliability of benefit estimates as an assessment of the viability of benefit transfer methods under circumstances that may guarantee successful benefit transfer. He introduces two types of reliability tests given that successful benefit transfer relies heavily on the similarities of physical and population characteristics between the study site and the policy site. The *split half sample tests* of reliability actually use identical physical and population characteristics of the hypothetical study and policy sites to evaluate the similarity of estimation results from two samples of the same population taken at the same time. If estimated measures of the central tendency (mean or median) and the demand or benefit function across the two samples are acceptably similar, the viability of a hypothetical benefit transfer in these most desirable circumstances (identical physical and population characteristics) may be guaranteed. The *test-retest method* of assessing the temporal reliability compares estimation results from two samples of the same population taken at different times. By evaluating the similarity of benefit estimates across the two samples taken at different times, the test-retest method may also provide an assessment of the viability and temporal stability of benefit transfer. Although these reliability tests involve purely hypothetical benefit transfer, there would be little potential for benefit transfer if these desirable circumstances do not provide reliable results.

The reliability of benefit estimates from various non-market valuation techniques depends on different factors due to the uniqueness of each valuation technique. For example, the selection of a functional form along with the specification of the independent variables for this function determines the reliability of the resulting value estimates in the travel cost method (Kling 1988), while the reliability of the hedonic price method depends on the selection of the dependent variable (e.g., housing price) and the hedonic functional form (Crooper, Deck, and McConnell 1988). The realism of the scenario, the use of sufficiently large sample sizes, and the robustness of statistical techniques determine the reliability of value estimates from contingent valuation method (Mitchell and Carson 1989; Hanemann 1994).

2.3.5 Feasibility of Benefit Transfer

The increasing demand for valid and reliable benefit transfer value estimates is derived from the increasing demand for non-market valuation studies by the federal or local government (public policy and project evaluations) and legal profession (natural resource damage assessment claims) in the United States (O'Doherty 1995). Since there are constraints on time and budget in conducting full-scale non-market valuation research, speedy and inexpensive yet acceptably accurate alternative method of estimating non-market value would be necessary in increasingly many occasions. As a result of the increasing demand for non-market valuations under time and financial constraints, the focus on the feasibility of benefit transfer along with the validity and reliability has been increasing. The required level of accuracy and scrutiny could be diverse depending on the circumstances of decision settings and the ultimate usage of benefit transfer estimates (Brookshire 1992; Desvousges, Dunford, and Mathews 1992). When the estimated benefits are considerably greater than the estimated costs associated with a proposed policy or program, the level of scrutiny required may not be very high. In addition, it would be more appropriate and safer to adopt conservative measures (the lower confidence limit) of benefit estimates against the upper confidence limit of cost estimates (O'Doherty 1995). The evaluation of the feasibility of benefit transfer methodologies is primarily based on the assessment of the degree of required accuracy and scrutiny along with the degree of fulfillment of necessary conditions for successful benefit transfer.

Brookshire (1992) proposes a spectrum of the required level of accuracy based on different purposes of performing benefit transfer as illustrated in Figure 2.1. To simply obtain some basic knowledge or facts in an attempt to determine the scope of proposed policies or programs, a relatively low level of accuracy may be sufficient for benefit transfer applications. Benefit transfer could serve as a useful screening tool for guiding the design of an original valuation study in the preliminary process of policy formulation. The required level of accuracy in this preliminary decision setting is not as high as the required level of accuracy for policy recommendations. To assist policy makers to evaluate alternative actions in the decision process for various environmental policies and programs, a higher level of accuracy should be guaranteed although benefit cost analysis may only need to determine whether expected benefits are greater than expected costs in some cases. In the context of natural resource damage assessment (NRDA) litigation and some public policies or programs based on externality costing, actual estimates of non-market value are required with the highest standard of accuracy to determine compensable damages in NRDA cases and to calculate externality costs for environmental policies (e.g., Pigouvian tax) designed to equate marginal social costs and marginal social benefits. The degree of required accuracy is closely related to the costs of making an inaccurate decision based on benefit transfer results. As potential social costs of inappropriate policy decisions resulting from inaccurate benefit transfer estimates increase, the level of required accuracy for benefit transfer estimates would necessarily increase. If the costs associated with a wrong decision appear to be too significant or irreversible, an original valuation study may be better conducted since potential benefits of using benefit transfer may not be large enough to offset potential costs.

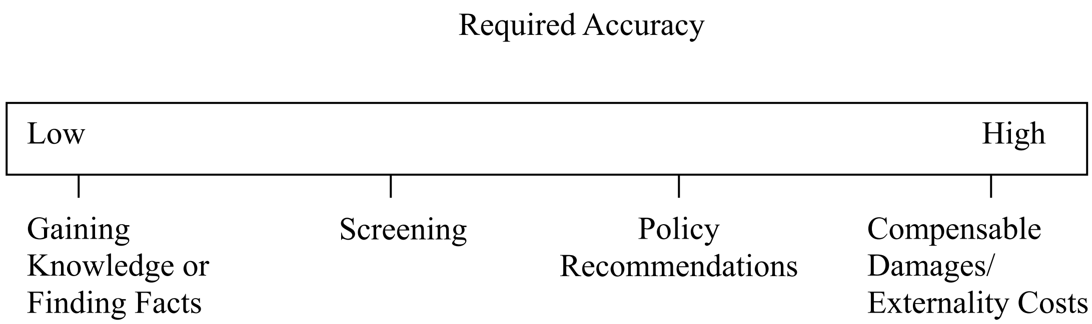

Figure 2.1: Spectrum of Accuracy Requirements
Source: Brookshire (1992)

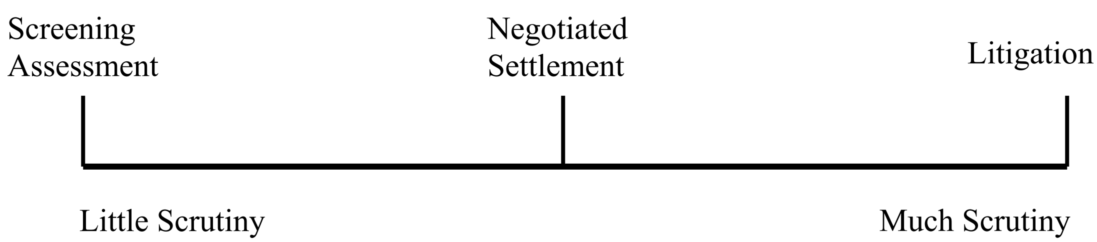

Figure 2.2: Continuum of Valuation Scrutiny in a NRDA Context
Source: Desvouges, Dunford, and Mathews (1992)

A continuum of valuation scrutiny in a NRDA context introduced by Desvouges, Dunford, and Mathews (1992) is illustrated in Figure 2.2. Their scrutiny continuum includes initial screening assessment of natural resource damages, negotiated settlement, and litigation that require different levels of scrutiny. Taking a more optimistic attitude, compared to Brookshire (1992), toward the potential advantages of benefit transfer in the NRDA procedures in the U.S., they report that the use of benefit transfer estimates is strongly supported for an initial screening assessment of damages or negotiated settlement. In their group discussion on the use of benefit transfer estimates in NRDA cases, participants support the use of benefit transfer for the initial process of NRDA or for a negotiated settlement that involves relatively little scrutiny; however, they don't support the adoption of benefit transfer estimates in a litigation context that requires much higher level of scrutiny. Because of high level of potential gains and losses with high level of required scrutiny (e.g., Exxon oil spills in the Valdes, AK and Arthur Kill in1989 and 1990), potential errors of benefit transfer estimates would not be defensible in a court when litigation is involved.

Based on the two continuums of required accuracy and scrutiny levels illustrated in Figures 2.1 and 2.2 with the degree of fulfillment of necessary conditions for successful benefit transfer (section 2.3.1), Figure 2.3 suggests a guideline on the recommended use of benefit transfer. Where the required levels of accuracy and scrutiny are low with high degree of fulfillment of necessary conditions for successful benefit transfer, the application of benefit transfer is most highly recommended. The application of benefit transfer methods, however, is never recommended where the required levels of accuracy and scrutiny are high with low degree of fulfillment of necessary conditions for successful benefit transfer. In other intermediate situations, a rather cautious approach is recommended in deciding whether to use benefit transfer even in a limited context.

Required Accuracy/Scrutiny

Degree of Fulfillment of Necessary Conditions for Successful Benefit Transfer		Low	High
	Low	Possibly Limited Use	**Never Recommended**
	High	**Highly Recommended**	Limited Use

Figure 2.3: The Recommended Use of Benefit Transfer
Source: O'Doherty (1995)

2.4 Benefit Transfer Methods

As discussed earlier, there are two broadly defined approaches of benefit transfer: *value transfer* and *function transfer*. These two benefit transfer approaches are illustrated in Figure 2.4. Value transfer techniques transfer a single benefit estimate from the study site or the measure of central tendency (e.g., mean or median value) for several benefit estimates from the study site(s), while function transfer techniques transfer an entire demand or benefit function from the study site. In both approaches, study site benefit estimates (value transfer) and a benefit or demand function (function transfer) are adapted to the differences between the study site and the policy site before being transferred to the policy site.

The simplest technique of benefit transfer is to estimate aggregate economic value of recreation activities or environmental resources (e.g., recreational fishing or water quality improvement) at the policy site by simply taking a single mean unit value (consumer surplus per trip or per day) or the average of several mean unit values from study site estimate(s), and multiplying this by the number of the affected population and possibly by their estimated recreation trips at the policy site. The main underlying assumption with these value transfer methods is that the change in welfare for an average individual at the study site would be equivalent to the change in welfare for an average individual at the policy site. If the physical characteristics of the policy and study sites, the socio-economic profiles of relevant populations, or the nature of environmental resources or changes being evaluated is different, direct transfer of benefit estimates could be, however, misleading unless benefit transfer estimates are carefully calibrated to reflect these differences. Study site estimates could be adjusted to reflect the differences between the study site and the policy site before being transferred to the policy site primarily based on professional judgments of researchers. Value transfer methods are only recommended where the benefit function for the study site or the mean values of its independent variables for the policy site are not available (Azqueta and Touza 2000).

For function transfer methods, the entire demand or benefit function estimated for the study site is used along with the relevant, primary and/or secondary data at the policy site for the variables included in the estimated demand or benefit function. Adapted policy site benefit transfer estimates can be predicted by inserting the mean values of the study site function's variables available at the policy site into the benefit function estimated at the study site. Function transfer assumes that underlying behavioral relationship between a recreation trip and the variables representing site and population characteristics is identical, and adjusts to the differences in these variables between the policy and study sites. Compared to value transfer benefit estimates, function transfer benefit estimates tend to be less biased from primary study value estimates available at the policy site (assumed true value for the purpose of convergent validity test) possibly due to more systematic adjustment by inserting available policy site mean values into the estimated study site benefit or demand function.

As more original valuation studies become available, meta-analysis function transfer technique estimates a regression equation of benefit estimates from various original studies with the characteristics of recreation activity, site, population, and valuation methodology as independent variables. Economic value needed for the policy site can be predicted by inserting the policy site values of relevant explanatory variables (e.g., summary statistics of population and site characteristics, the type of recreation activity, and the geographic location) into a meta-analysis benefit function holding other detailed valuation methodology variables at their sample mean values, national average values for instance. The national or regional average value could also be predicted by setting all independent variables of a meta-analysis benefit function at their national mean values except for the relevant activity and region variables. A meta-analysis benefit function can help us better understand the influence of valuation methodologies (revealed or stated preference methods) and other study-specific factors (detailed strategies for each valuation method) on benefit estimates by systematically adapting original valuation studies to the differences in these detailed research methods. However, many variables may need to be standardized for consistency, especially the variables that represent different aspects of the quality of environmental resources since original valuation studies were not designed to be pooled together for meta-regression analysis.

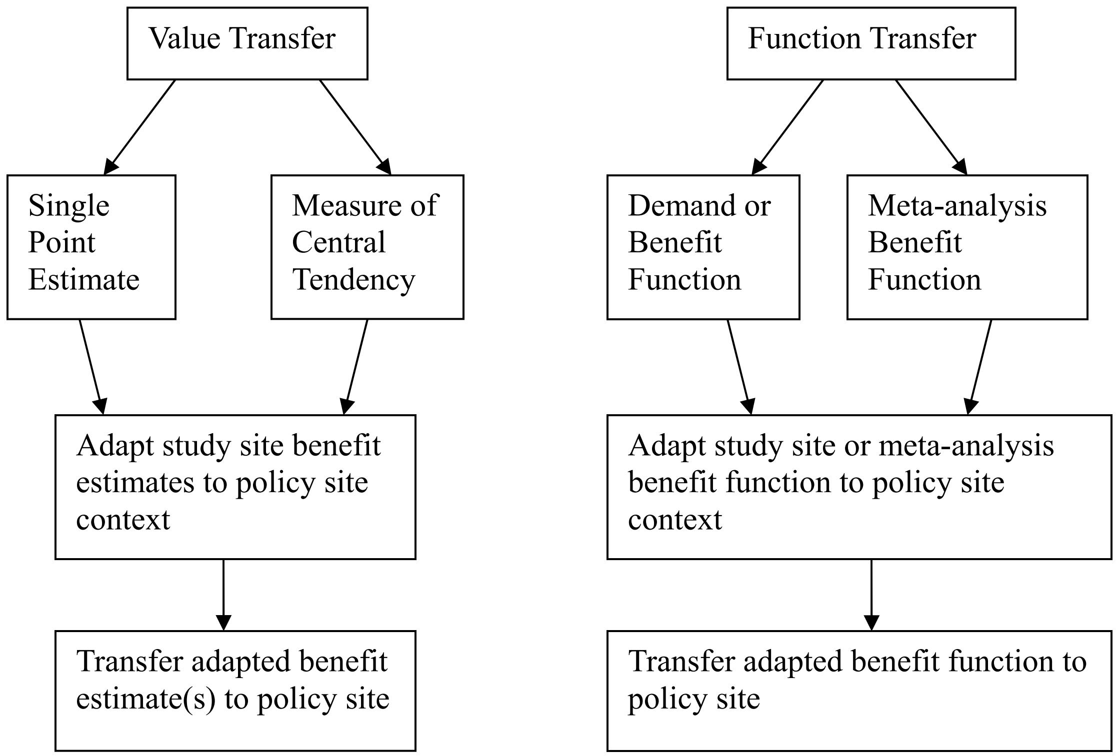

Figure 2.4: Two Benefit Transfer Approaches
Source: Rosenberger and Loomis (2001)

2.5 Benefit Transfer Procedures

A contribution to the development of a generally accepted protocol or systematic procedures for applying benefit transfer methods has been a major goal of many previous studies on benefit transfer although it appears that appropriate benefit transfer procedures may be dependent, at least to some extent, on the specific circumstances under which benefit transfer is applied. Several studies (Boyle and Bergstrom 1992; Kask and Shogren 1994; Desvousges, Johnson, and Banzhaf 98; Azqueta and Touza 2000; Brouwer 2000; Rosenberger and Loomis 2001, 2003) suggest basic steps to be followed in carrying out benefit transfer. To summarize benefit transfer procedures recommended by previous studies, five basic steps in applying benefit transfer methods are illustrated in Figure 2.5.

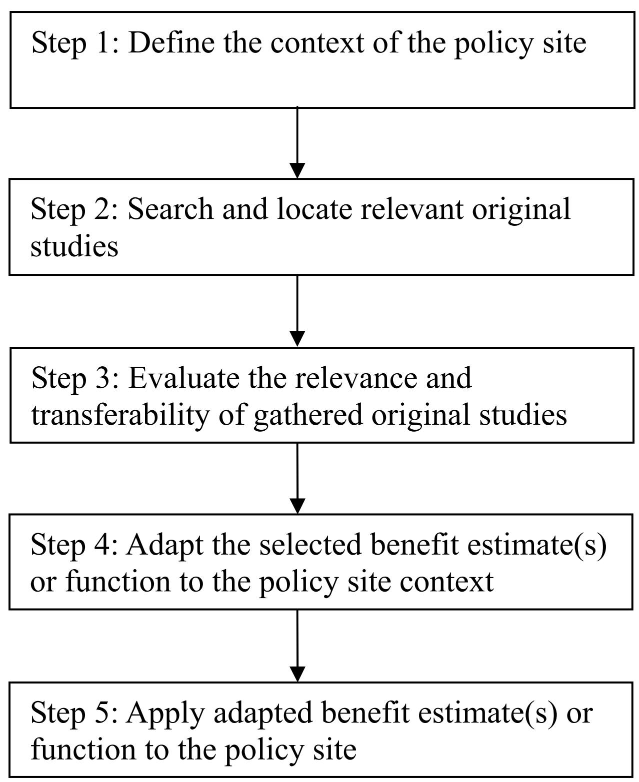

Figure 2.5: Basic Steps in Benefit Transfer Procedure

The first step is to *define the context of the policy site* to which benefit estimates are to be transferred. The environmental resource and its expected change to be evaluated should be carefully specified. Socio-economic characteristics (e.g., income, education, age, and gender) of the affected population and physical characteristics of the policy site (e.g., environmental quality, geographic location, and accessibility conditions) that can influence benefit estimates are thoroughly defined in this first step. Kask and Shogren (1994) recommend that the purpose and the level of required accuracy of benefit transfer estimates should be also determined in this step.

The second step is to *search and located relevant original studies* from which benefit estimates might be transferred. A thorough literature search should be conducted by reviewing relevant journal articles, working papers, books, unpublished government reports, conference papers, and doctoral theses to locate and gather all the relevant original valuation studies that can be potentially applied to the policy site. As more and greater diversity of original non-market valuation studies become available coupled with the increasing demand for benefit transfer methods, internet-based large databases of original non-market valuation studies can be made available. These nation or world wide databases may provide actual benefit estimates and estimation methodologies used, so that the relevance and applicability of existing valuation studies to benefit transfer can be evaluated systematically. The Environmental Valuation Reference Inventory (EVRI) by Environment Canada and the environmental valuation database (ENVALUE) by the Environmental Protection Agency of New South Wales are two general environmental valuation databases currently available. The Beneficial Use Values Database (BUVD) of water resource by the University of California, Davis and the sport fishing values database by the U.S. Fish and Wildlife Service are more specialized databases available.

The third step is to *evaluate the relevance and transferability of gathered original studies*. Various original valuation studies obtained from the literature search conducted in the second step should be carefully evaluated to determine whether they are relevant and transferable to the context of the policy site. Both Boyle and Bergstrom (1992) and Desvousges, Naughton, and Parsons (1992) suggest several criteria to select among available original valuation studies for benefit transfer based on the similarities of the environmental resource and its change being evaluated, the characteristics of sites and affected populations, and theoretical welfare measures between the study site and the policy site. Along with the similarities of these aspects, the adequacy of the data, economic methods, and empirical techniques used in the gathered original valuation studies should also be evaluated.

The fourth step is to *adapt the selected benefit estimate(s) or function to the policy site context*. It is most likely that the characteristics of sites and affected populations and the nature of environmental changes at the study and policy sites are not identical, necessitating adjustment processes for the selected study site benefit estimates or benefit function to better reflect the differences in these attributes. More systematic adjustments of study site benefit estimates or benefit function could provide more reliable (smaller variance) and valid (smaller bias) benefit transfer estimates for the policy site. Usually in function transfer methods, some primary or secondary data collection at the policy site may be necessary to gather relevant summary data for the variables included in the estimated study site benefit function before attempting to predict the policy site benefit estimates by transferring this study site function.

The final step is to *apply adapted benefit estimate(s) or function to the policy site*. This is the stage where actual benefit transfer estimates for the policy site are calculated. Benefit transfer could be carried out as value transfer (the transfer of a single point estimate or the average of several estimates), function transfer (the transfer of an entire demand or benefit function), or meta-analysis function transfer (the transfer of a meta-analysis benefit function) with systematic adjustment processes using the information obtained from the original valuation studies. These adapted or adaptively predicted benefit transfer estimates for the policy site could be aggregated by multiplying unit value (e.g., consumer surplus per trip) by the number of relevant units (e.g., total number of affected population and predicted recreation trips per person) to yield a measure of total benefit or cost at the policy site.

2.6 Modeling Benefit Transfer

Rosenberger and Loomis (2003) comprehensively illustrate the process of modeling benefit transfer methods: value transfer, function transfer, and meta-analysis function transfer. Following their notations, let V_S (V_{Si}) and V_P (V_{Pj}) represent the estimated and needed measures of environmental resource value for the study site (study site *i*) and the policy site (policy site *j*) respectively. We can estimate the needed measure (V_{Pj}) for the policy site *j* by adaptively transferring study site measure(s) (V_{Si}) through the transfer of benefit estimates, a benefit function, or a meta-analysis benefit function, then study site value becomes benefit transfer value (V_{BT}) when applied to the policy site *j* (V_{Si} => V_{BT}). Modeling benefit transfer methods is about how V_S (estimated measure for the study site) can be used to estimate V_P (needed measure for the policy site).

2.6.1 Value Transfer Methods

Value transfer can be defined as the direct or adaptive application of a single or several study site benefit estimates to the policy site. There are two approaches (Figure 2.4) to conducting value transfer: point estimate transfer and average value transfer.

Point Estimate Transfer

Point estimate transfer is based on a single measure or possibly a range of point estimates obtained from relevant primary studies. The measure(s) of study site value (V_{Si}) under the context of the study site *i* (X_{Si}) can be used to predict the needed policy site value (V_{Pj}) under the context of the policy site *j* (X_{Pj}):

$$V_{Pj}|X_{Pj} = V_{BT} = V_{Si}|X_{Si} \quad (2.1)$$

where V_{BT} is benefit transfer value for the policy site. When a range of benefit estimates is transferred, a confidence interval around transferred point estimates can be constructed if possible by proving bounds on the estimated policy site value. It must be noted that all study site benefit estimates should be expressed in a comparable index such as consumer surplus per activity day (or per trip) per person adjusted for inflation before being transferred to the policy site.

Average Value Transfer

Average value transfer is based on the measure of central tendency for benefit estimates obtained from relevant primary studies. This approach is similar to point estimate transfer except for the use of an average or other measure of central tendency for relevant and transferable study site value estimates. A mean, median, or other measure of central tendency based on all or a subset of relevant study site benefit estimates is used to predict the needed policy site value. This approach is defined as

(2.2) $V_{Pj}|X_{Pj} = V_{BT} = \overline{Vs \mid Xs}$

where V_{Pj} is the needed value for the policy site *j* under the policy site context (X_{Pj}) and $\overline{Vs \mid Xs}$ is the measure of central tendency for all or a subset of study site measures under each study site's context. Again, all benefit measures used to calculate the measure of central tendency should be adjusted to a common unit relevant to the policy site. Since the average (e.g., mean) value could be affected by extremely large or small "outlier" benefit estimates, only a subset of estimates may have to be used especially when there are a small number of relevant benefit estimates.

Convergent Validity Test

To empirically test the convergent validity (unbiasedness) of benefit transfer estimates, we need to know the "true value" of the environmental resource at the policy site. The only practical way of performing a convergent validity test is to compare benefit transfer estimates (V_{BT}) with primary study estimates available at the policy site (V_P) that are considered as the best approximation of the true value. That is, we actually assume that original study estimates at the policy site are the true value, and compare them with benefit transfer estimates for the policy site. The error associated with benefit transfer can be defined as

$$(2.3) \quad \delta_{BT} = (V_{BT} - V_P)/V_P$$

where V_{BT} is the transferred value from the study site and V_P is the assumed true value from the primary study at the policy site. The magnitude of benefit transfer error (the percentage difference between the assumed true value and transferred value) depends primarily on the degree of correspondence between the study site and the policy site, including the characteristics of sites, affected populations, and the environmental resource and its change.

2.6.2 Function Transfer Methods

Function transfer methods involve the transfer of functions or statistical models that relate benefit measures to the study site characteristics, including a demand or benefit function and meta-analysis benefit function (Figure 2.4). Function transfer methods are generally considered as providing more accurate (smaller benefit transfer error) estimates than value transfer methods because function transfer methods are more systematically tailored to the differences between the study site and the policy site.

Demand or Benefit Function Transfer

Demand or benefit function transfer is based on the assumption that the study site value (V_{Si}) can be expressed as a function of the explanatory variables that represent the study site context (X_{Si}) (e.g., physical features of the site and socio-economic characteristics of the relevant population):

(2.4) $V_{Si} = f_S(X_{Si})$.

By requiring a higher degree of correspondence (similar X_{Si} and X_{Pj}), value transfer methods are less adaptive to significant differences between the study site and the policy site. The transfer of an entire demand or benefit function to the policy site should increase the accuracy of benefit transfer estimates because a demand or benefit function could be tailored to the specific context of the policy site such as location, physical features, climate, and socio-economic variables as long as these variables are included in the study site function. The demand or benefit function transfer is defined as

$$(2.5) \quad V_{Pj} = V_{BT} = f_{S|P}(X_{Pj})$$

where the needed value for the policy site (V_{Pj}) is derived from the study site demand or benefit function adapted to the context of the policy site ($f_{S|P}(X_{Pj})$).

The demand or benefit function itself from the study site may have to be adapted ($f_{S|P}(.)$) to the measured differences between study site variables (X_{Si}) and policy site variables (X_{Pj}) before being transferred since it may be the case that study site and policy site variables are not directly comparable. For example, a study site variable may be a continuous variable (years of education), but the same variable for the policy site could be a discrete variable (a categorical variable based on the degree acquired). In this case, the effect (a regression coefficient from the study site function) of this variable on the benefit measure obtained from the study site should be adjusted to reflect different formats of the same socio-economic variable, education. By adapting the study site function to the policy site context, demand or benefit function transfer provides tailored benefit estimates for the policy site under the assumption of the identical statistical relationship (regression coefficients) between benefit measure and the characteristics of site and relevant population at the study and policy sites.

Meta-analysis Function Transfer

Meta-analysis benefit function transfer is based on the statistical summarization of the relationship between benefit measure and the detailed characteristics of original valuation studies. Meta-regression analysis typically uses summary statistics data from the existing valuation studies such as benefit measures, site and population characteristics, environmental commodity, and detailed valuation methodologies. Meta-analysis benefit function transfer is defined as

(2.6) $V_S = f_{MA}(\overline{X}_S, M_S)$ (Meta-analysis benefit function)

(2.7) $V_{Pj} = V_{BT} = f_{MA|P}(\overline{X}_{Pj}, \overline{Zs}, \overline{Ms})$ (Transferred function)

where V_S is a vector of benefit measures from original studies; f_{MA} is a meta-analysis regression function; $\overline{X}_S$ is a vector of summary statistics for socio-economic and site characteristics variables (e.g., income, age, education, gender, geographic location, accessibility, and site quality); M_S is a vector of methodological variables (e.g., valuation method, modeling format, and functional form); $\overline{X}_{Pj}$ is a vector of socio-economic and site characteristics variables relevant to the policy site *j*; and $\overline{Zs}$ and $\overline{Ms}$ are the vectors of irrelevant site characteristics and methodology variables at their sample mean values (e.g., national or regional average) respectively.

A meta-analysis benefit function (equation (2.6)) traditionally tries to explain the influence of methodological (M_S) and study-specific ($\overline{X}_S$) factors on research outcomes (V_S) and to provide meaningful summaries and synthesis of past research effort (Stanley and Jarrell 1989). A meta-analysis can statistically explain the variation of benefit estimates across many valuation studies with different valuation method, survey mode, geographic location, time, and other study-specific features. These methodological and study-specific attributes cannot be independent variables in individual valuation studies; therefore, meta-regression analysis may only be able to identify individual effects of these variables on benefit estimates. Benefit transfer estimates tailored to the policy site can be obtained by adapting the meta-analysis benefit function to the specific characteristics of the policy site (equation (2.7)). All the irrelevant or unavailable variables are set to their sample mean values, and insert relevant policy site variables into the estimated meta-analysis benefit function. Estimated meta-regression coefficients may have to be adjusted ($f_{MA|P}(.)$) to reflect different formats of the same variable between the study site and the policy site, let alone possible adjustment of the same variable from multiple study sites to make it more amenable to meta-regression analysis.

Convergent Validity Test

Existing empirical studies trying to compare the validity of value transfer and function transfer estimates using equation (2.3) generally suggest that function transfer estimates are more accurate than value transfer estimates (Loomis 1992; Parsons and Kealy 1994; Loomis et al. 1995; Downing and Ozuna 1995; Kirchhoff, Colby, and LaFrance 1997; Brouwer and Spaninks 1999; Rosenberger and Loomis 2001). The improved accuracy of function transfer methods is probably due to the ability of adapting benefit functions to specific attributes of the policy site. Some studies suggest that meta-analysis benefit function transfer performs better than demand or benefit function transfer (Brouwer and Spaninks 1999; Rosenberger and Phipps 2001; VandenBerg, Poe, and Powell 2001). Rosenberger and Phipps (2001) demonstrate that study-specific site characteristics that are invariant within an individual study, but vary across different studies, can explain much of the error associated with benefit transfer methods. However, these effects could be controlled for if meta-analysis benefit function transfer is performed, thus providing the increased accuracy of benefit transfer estimates.

2.7 Benefit Transfer Application: Marine Recreational Fishing

The importance of and need for efficient and effective management programs for recreational fisheries as a renewable resource have been recognized to accomplish an economically and biologically sustainable level of harvest (catch and keep). From 15 to 17 million marine recreational anglers took over 86 million fishing trips and harvested over 189 million fish weighing almost 266 million pounds (over 254 million fish were caught and released) in 2001. Thus, marine recreational fishing could have significant economic impacts on coastal regions and the areas where market goods related to marine recreational fishing are produced, let alone a large impact on available fish stocks (the MRFSS). To develop fishery management policies and evaluate the impacts of resulting regulations on marine recreational anglers and fisheries, the NMFS collects data on the number and socio-economic characteristics of marine recreational anglers; total number of fishing trips by them; and the number, size composition, and weight of recreational harvest through the MRFSS combined with the AMES.

The method of function transfer is applied to evaluate how well benefit transfer performs in the estimation of non-market recreational value associated with marine recreational fishing in the coastal areas of the U.S. using two original valuation studies with a high level of correspondence in many aspects (Table 2.1). Using a two-stage nested random utility model (NRUM) for single day marine recreational fishing trips, both Hicks et al. (1999) and Haab, Whitehead, and McConnell (2001) estimate the economic value associated with access to county-level zone fishing sites (willingness to pay (WTP) for the opportunity of marine recreational fishing in a particular area) and a one unit increase in five-year historic harvest rate (willingness to pay for the better opportunity of catching fish) using the Northeast (NE) 1994 and Southeast (SE) 1997 MRFSS-AMES data respectively. Both NE and SE coastal regions in the U.S. are considered as potential candidates for both the study site and the policy site in carrying out function transfer. The results of original estimations (NE 1994 and SE 1997) are compared with the results of benefit transfer estimations to empirically assess the convergent validity (equation (2.3)) of benefit function transfer estimates in a marine recreational fishing environment with MRFSS data.

	Hicks et al. (1999)	***Haab, Whitehead, and McConnell (2001)***
Recreation activity	Saltwater sport fishing: one day trip	Saltwater sport fishing: one day trip
Data	MRFSS-AMES: 1994 Northeast	MRFSS-AMES: 1997 Southeast
States included	VA, MD, DE, NJ, NY, CT, RI, MA, NH, & ME	NC, SC, GA, FL, AL, MS, & LA
Estimation technique	two-stage nested random utility model (NRUM)	two-stage nested random utility model (NRUM)
Welfare measures	WTP for site access to a state across waves[1] (3~6) & for one unit ↑ in historic harvest rate by state and 4 species groups	WTP for site access to a state across waves (2~6) & for one unit ↑ in historic harvest rate by state and 4 species groups
Choice set	3 fishing modes-5 target species & 63 county-level zone sites	3 fishing modes-5 target species & 70 county-level zone sites
Explanatory variables of indirect utility function	Trip cost & time, # interview sites in a county zone, & site-specific historic harvest per trip for species group	Trip cost & time, # interview sites in a county zone, & site-specific historic harvest per trip for species group

1. A wave is a two-month period: Jan/Feb (wave1) ~ Nov/Dec (wave6).

Table 2.1: Summary of Two Original Valuation Studies

2.7.1 Two Primary Studies of Marine Recreational Fishing Value

Using the same data source although in different years and regions, two studies (Hicks et al. 1999; Haab, Whitehead, and McConnell 2001) attempt to evaluate the identical welfare measures (WTP for site access and site quality increase) of single-day saltwater sport fishing trips with the same estimation technique (NRUM). Hicks et al. (1999) examine the economic value associated with marine recreational fishing in the Northeast region (ten coastal states from Virginia through Maine) of the United States in 1994, while Haab, Whitehead, and McConnell (2001) examine the same value in the Southeast region (seven coastal states from North Carolina through Louisiana) in 1997. Mode/species-site choice model of marine recreational fishing behavior is estimated with a two-stage nested random utility model assuming that the angler first determines one of 15 possible mode-species combinations from three fishing modes (private/rental boat fishing, charter/party boat fishing, and shore fishing) and five species groups (big game, small game, bottom fish, flat fish, and no/other target), and then chooses a specific fishing site conditional on the choice of mode-species combination (Figure 2.6). The only difference in the choice structure is the number of alternative county-level fishing zones at the second stage of the decision process: 63 and 70 alternative zone sites in the Northeast 1994 and Southeast 1997 studies respectively.

The explanatory variables used in the estimation of a two-stage NRUM include travel cost (explicit and opportunity costs), travel time (if anglers respond that they don’t lose any income due to fishing trips), the number of interview sites in an aggregated county-level zone (correction for aggregation bias), and historic harvest rate per trip by wave, species, mode, and site (site quality measure). In the specification of mode-species specific inclusive values that will determine the probability of choosing a mode-species combination, Haab, Whitehead, and McConnell (2001) allow the inclusive value parameter to differ between four targeted species (big game, small game, bottom fish, and flat fish) and other non-targeted species by recognizing possibly different substitution patterns, while Hicks et al. (1999) use a single inclusive value parameter for all targeted and non-targeted species.

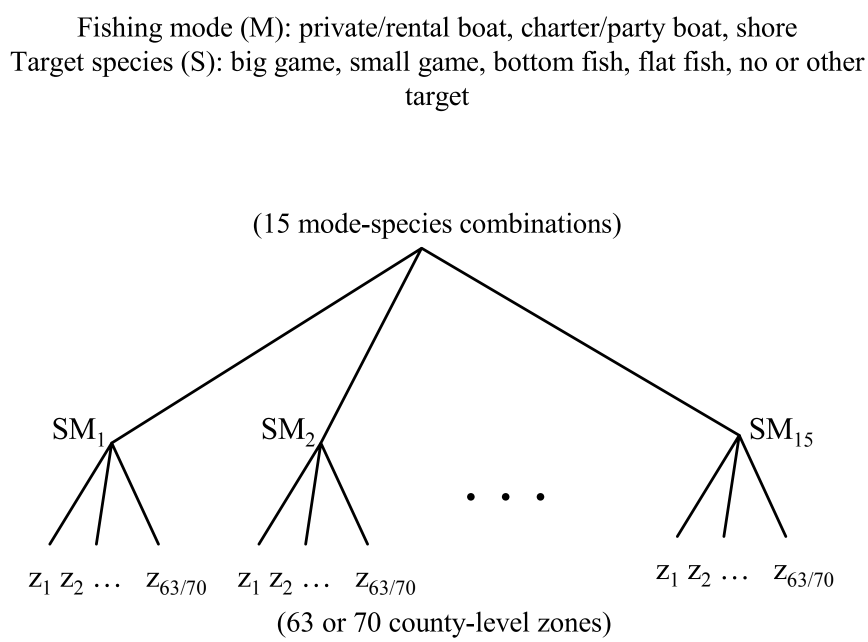

Figure 2.6: The Choice Structure in a Two-stage NRUM

Welfare measures estimated in both studies are the economic value associated with access to fisheries and the quality of fishing experience. They estimate the value of access to fisheries in each state as the mean willingness to pay per trip for site access to a particular state across waves assuming all fishing sites in the state and wave are closed under current choices of all anglers. As a proxy for the quality of fishing experience in each site, they use the average harvest per trip by averaging actual kept catches by wave, fishing mode, target species, and site over the past five-year period (historic harvest rate). To measure the marginal willingness to pay for a one-unit increase in historic harvest rate by state and species group, historic harvest rate per trip for each species group is increased by one unit at all fishing sites in a particular state for all anglers.

2.7.2 Original Estimation Model

A marine recreational angler is assumed to jointly choose target species and fishing mode at the first stage, and then choose among mutually exclusive fishing sites based on their attributes at the second stage (two-stage mode/species-site choice model). If we denote alternative sites and mode-species combinations with j (1,…,63 (NE 1994) or 70 (SE 1997)) and sm (1,…,15) respectively, an indirect utility function of an arbitrary angler can be written as (following Haab, Whitehead, and McConnell 2001)

$$(2.8) \quad v_{jsm} = \beta_1 c_j + \beta_2 tt_j + \gamma_1 logM_j + \sum_{s=1}^{5} \gamma_{2s} d_s \sqrt{q_{jsm}} + \varepsilon_{jsm}$$

where ν_{jsm} is the deterministic utility for site j and mode/species combination sm, c_j is the travel cost to site j, tt_j is the travel time for those who cannot value the travel-time at the wage rate, M_j is the number of intercept sites in the aggregated county level zone, q_{jsm} is five-year historic harvest rate for species s through mode m at site j, d_s is a species dummy variable, and ε_{jsm} is a generalized extreme value random error term.

The probability of choosing site j conditional on mode/species choice sm, mode/species-specific inclusive value, and probability of choosing mode/species combination sm are

$$(2.9)\quad \text{Prob}(j|sm) = \exp[(\beta_1 c_j + \beta_2 tt_j + \gamma_1 logM_j + \sum_{s=1}^{5}\gamma_{2s} d_s \sqrt{q_{jsm}})/\theta s]$$

$$/\sum\nolimits_h \exp[(\beta_1 c_h + \beta_2 tt_h + \gamma_1 logM_h + \sum_{s=1}^{5}\gamma_{2s} d_s \sqrt{q_{hsm}})/\theta s]$$

$$(2.10)\quad I_{sm} = \ln(\sum\nolimits_h \exp[(\beta_1 c_h + \beta_2 tt_h + \gamma_1 logM_h + \sum_{s=1}^{5}\gamma_{2s} d_s \sqrt{q_{hsm}})/\theta s])$$

$$(2.11)\quad \text{Prob}(sm) = \exp(\theta s\, I_{sm}) / \sum\nolimits_n \exp(\theta s\, I_n)$$

where θs is a species-specific inclusive value parameter and I_{sm} is the mode/species-specific inclusive value. The estimation of the second stage site choice decision (equation (2.9)) yields the estimates of $(\boldsymbol{\beta}, \boldsymbol{\gamma})/\theta s$, and then the inclusive values (equation (2.10)) can be calculated using these parameter estimates for the estimation of the first stage mode-species choice decision (equation (2.11)). In both NE 1994 and SE 1997 data, the inclusive value parameters for the four targeted species groups are assumed to be the same (θ_T), and the inclusive value parameter for the non-targeted species is assumed to be different (θ_{NT}) since the pattern of substitution between sites is expected to differ for those who do not target a particular species. Hicks et al. (1999), however, don't allow the inclusive value parameter for the anglers with no target species to differ.

The standard welfare measure from a nested logit random utility recreational fishing model that is linear in travel cost compares the expected maximum utility after policy change (V^1) with a baseline level of the expected maximum utility (V^0), and then converts the difference into a money metric by normalizing with the marginal utility of income (β_1). Given the indirect utility function in equation (2.8), the expected maximum utility under policy situation z (V^z) is

$$(2.12)\quad V^z = \ln\left[\sum\nolimits_{tm}\left(\sum\nolimits_j \frac{v^z_{jtm}}{\theta_T}\right)^{\theta_T} + \sum\nolimits_{nm}\left(\sum\nolimits_j \frac{v^z_{jnm}}{\theta_{NT}}\right)^{\theta_{NT}}\right]$$

where the first summation is over the 12 mode/species combinations that contain targeted species groups, the third summation is over the 3 mode/species combinations with no target, and v^z_{jsm} (v^z_{jtm} or v^z_{jnm}) is the estimated indirect utility function evaluated at independent variable values under situation z.

It is possible to introduce a policy regime that changes the value of independent variables included in the indirect utility function. Two policy situations considered in the analysis are a closure of all fishing sites in a state during a particular wave and an increase in the historic harvest rate at all fishing sites in a state for each species group to measure the access value of fishing in the state for all anglers and the marginal willingness to pay for a one fish increase in the harvest rate at all sites respectively. In these cases, the expected maximum utility is adjusted by either eliminating the affected sites (j) or increasing harvest rates (q_{jsm}) from the corresponding summations in equation (2.12). The willingness to pay for a policy change or the welfare change from policy situation z = 0 to z = 1 (assuming welfare enhancing change) can be measured as

(2.13) $WTP = (V^0 - V^1)/\beta_1$

where V^0 is a baseline level of the expected maximum utility under situation 0, V^1 is the expected maximum utility after a policy change to situation 1, and β_1 is the estimate of travel cost coefficient obtained from the estimation of the second stage site choice decision (equation (2.9)).

2.7.3 Original Estimation Results: NE 1994 and SE 1997 MRFSS-AMES Data

The estimation results of a two-stage nested random utility model of marine recreational fishing using NE 1994 and SE 1997 data are presented in Table 2.2. In both NE and SE models, travel cost (explicit out of pocket costs + implicit opportunity cost of travel time) and travel time variables have a negative and significant effect on site choice as expected implying that trip-related expenses and opportunity cost of traveled time are inversely related to site choice probability. The number of available fishing sites included in an aggregated county zone positively influences the probability of choosing that zone in both models. All historic harvest rate variables that represent the quality of county level fishing zones have a positive effect on indirect utility with big game and flat fish species groups having the largest marginal utilities in NE 1994 and SE 1997 models respectively. In general, targeted species groups give anglers more marginal utilities than non-targeted species in both models suggesting that targeting a particular species could lead to more valuable fishing experience in a marine recreational fishing environment.

Inclusive value parameter estimates in both models generally support the appropriateness of a two-stage nested random utility model instead of a simple site choice model (unnested RUM). If an inclusive value parameter is close to one, a two-stage nested decision structure may not be appropriate. This may be the case for the anglers who do not target a particular species in SE 1997 model.

Variable	Definition	Coeff.	Std. Err.	Mean
	Northeast 1994			
Variable	*Definition*	*Coeff.*	*Std. Err.*	*Mean*
TRAVELC	Travel Cost	-0.028	0.002	193.14
TTIME	Travel Time	-0.9355	0.0432	10.55
LNM	Log(Number of NMFS Interview Sites in Aggregated Zone)	1.1507	0.032	3.14
MBIG	Square Root of Historic Harvest Rate: Big Game	1.1247	0.2803	0
MSMALL	Square Root of Historic Harvest Rate: Small Game	0.5229	0.0602	0.43
MBOTTOM	Square Root of Historic Harvest Rate: Bottom Fish	0.5625	0.0494	0.2
MFLAT	Square Root of Historic Harvest Rate: Flat Fish	0.7777	0.0789	0.17
MOTHER	Square Root of Historic Harvest Rate: Other	0.3349	0.0732	0.2
INC_T	Inclusive Value: Targeted Species	0.2473	0.0281	
INC_NT	Inclusive Value: Non-targeted Species	0.2387	0.0311	
	Southeast 1997			
Variable	*Definition*	*Coeff.*	*Std. Err.*	*Mean*
TRAVELC	Travel Cost	-0.0163	0.0008	330.31
TTIME	Travel Time	-0.5522	0.0136	22.53
LNM	Log(Number of NMFS Interview Sites in Aggregated Zone)	0.7941	0.0232	2.67
MBIG	Square Root of Historic Harvest Rate: Big Game	0.3551	0.157	0.02
MSMALL	Square Root of Historic Harvest Rate: Small Game	0.1804	0.0642	0.35
MBOTTOM	Square Root of Historic Harvest Rate: Bottom	0.0619	0.0523	0.09
MFLAT	Square Root of Historic Harvest Rate: Flat	0.4952	0.1773	0.01
MOTHER	Square Root of Historic Harvest Rate: Other	0.0098	0.0542	0.1
INC_T	Inclusive Value: Targeted Species	0.7326	0.1057	
INC_NT	Inclusive Value: Non-targeted Species	1.1162	0.1113	

Table 2.2: Two-stage Nested RUM Parameter Estimates

2.7.4 Original Welfare Estimation

Tables 2.3 and 2.4 present welfare estimates of the *mean value of access per trip* by state and two-month wave and *willingness to pay for a one fish increase in historic harvest rate per trip* by state and species group from NE 1994 and SE 1997 models respectively. At the first stage estimation (conditional site choice decision given mode-species combination) of a two-stage nested RUM, all parameter estimates are normalized by inclusive value parameter. Since we assume different inclusive value parameters for four targeted species groups (θ_T) and other non-targeted species group (θ_{NT}), a weighted inclusive value parameter is used to recover β_1 in equation (2.13). The proportions of anglers with any of four targeted species groups and non-targeted species group in the sample are used as corresponding weights.

In NE 1994 model (Table 2.3), Virginia (22% of total fishing trips) has the largest access value followed by New York, New Jersey, Maryland, Massachusetts, and Maine while New Hampshire (4.6% of total fishing trips) has the lowest access value among the Northeastern coastal states for all waves. There is no particular wave that generally has larger access value among all Northeastern states although the largest proportion (34.2%) of fishing trips occurs in wave 4 (July-August). Big game species group provides the largest gain per trip from a one fish increase in five-year historic harvest rate followed by flat fish and small game species groups while bottom fish species group provides the lowest gain per trip in all Northeastern states. For all targeted species groups, Maine and Maryland show relatively larger gains per trip from a one fish increase in harvest rate although variations are not very considerable.

In SE 1997 model (Table 2.4), Florida (60.26% of total fishing trips) has the largest access value followed by North Carolina and Louisiana while Alabama (3.2% of total fishing trips) has the lowest access value among the Southeastern coastal states for all waves. Again, there is no particular wave that has larger access value among all Southeastern states, and most fishing trips (23.83%) occur during the wave 3 (May-June) unlike the Northeastern coastal states with most fishing trips occurring during the wave 4 (July-August). In the Southeastern coastal states, flat fish species group provides the largest gain per trip from a one fish increase in historic harvest rate followed by big game and small game species groups while bottom fish species group provides the lowest gain per trip in all Southeastern coastal states. There is not any noticeable variation across states in gains per trip from a one fish increase in historic harvest rate of all targeted species groups.

In evaluating the mean values of access per trip by state, we should not add these values together across states to calculate the access value of multiple states since these values are calculated under the assumption that all of other alternative sites in other states are available to the angler. Simply adding these values together provides incorrect measures of access value of multiple or all states in the region. For the access value of multiple states in the region, all fishing sites in the considered states should be assumed simultaneously closed to calculate the access value of these closed states using equation (2.13). Table 2.4 actually shows the access value of some multi-state areas: Gulf of Mexico and South Atlantic areas. To accurately calculate the access value of whole region, survey data from another region should be combined to create multi-region data.

The Mean Value of Access Per Trip

State	All Waves	Wave 3	Wave 4	Wave 5	Wave 6
Connecticut	$5.31	$5.56	$5.70	$4.97	$4.58
Delaware	$2.42	$3.42	$2.78	$0.93	$2.50
Maine	$18.76	$20.29	$23.51	$21.83	$0.00
Maryland	$29.66	$32.86	$27.99	$35.94	$17.24
Massachusetts	$21.08	$22.31	$20.38	$25.50	$12.94
New Hampshire	$1.31	$1.91	$1.52	$1.21	$0.00
New Jersey	$34.90	$40.91	$33.19	$34.83	$28.89
New York	$58.93	$58.39	$56.12	$57.85	$68.19
Rhode Island	$9.91	$9.10	$10.35	$11.12	$8.18
Virginia	$117.46	$79.89	$95.29	$113.04	$238.64
Obs.	4897	1220	1675	1271	731

Willingness to Pay for a One Fish Increase in Historic Harvest Rate Per Trip

State	Obs.	Big Game	Small Game	Bottom Fish	Flat Fish
Connecticut	281	$21.85	$8.10	$5.92	$16.12
Delaware	190	$20.07	$7.38	$5.28	$15.19
Maine	273	$25.12	$9.55	$6.91	$21.59
Maryland	501	$25.67	$9.35	$6.52	$20.50
Massachusetts	529	$22.29	$7.74	$5.55	$16.03
New Hampshire	225	$22.83	$8.07	$5.77	$17.30
New Jersey	793	$18.15	$6.54	$4.71	$12.96
New York	678	$17.67	$5.81	$4.50	$12.00
Rhode Island	349	$20.70	$7.50	$5.41	$15.73
Virginia	1078	$16.27	$5.72	$4.76	$12.05
All States	4897	$19.96	$7.10	$5.28	$14.88

Table 2.3: Welfare Estimates from Northeast 1994 MRFSS-AMES Data

The Mean Value of Access Per Trip						
State	All Waves	Wave 2	Wave 3	Wave 4	Wave 5	Wave 6
Florida (All)	$300.12	$351.54	$287.54	$299.89	$270.55	$306.23
Florida West (Gulf)	$60.66	$74.53	$58.09	$56.23	$58.15	$59.10
Florida East (SA)	$16.33	$17.01	$13.81	$16.84	$15.51	$19.23
Georgia	$3.41	$1.17	$5.10	$4.45	$3.35	$2.40
N. Carolina	$37.19	$21.74	$39.61	$38.02	$49.58	$32.44
S. Carolina	$9.93	$10.02	$8.07	$9.37	$12.12	$10.12
Louisiana	$16.58	$12.23	$16.81	$19.34	$16.41	$17.61
Mississippi	$4.87	$4.61	$4.64	$4.64	$5.41	$4.96
Alabama	$2.09	$2.37	$2.53	$1.85	$1.61	$2.07
Gulf Coast	$113.42	$118.61	$109.15	$113.93	$114.82	$112.28
S. Atlantic	$162.37	$112.10	$168.07	$161.58	$201.10	$154.29
Obs.	6379	1039	1520	1115	1417	1288

Willingness to Pay for a One Fish Increase in Historic Harvest Rate Per Trip					
State	Obs.	Big Game	Small Game	Bottom Fish	Flat Fish
Alabama	206	$20.17	$9.79	$3.32	$27.78
Florida East (SA)	1398	$20.36	$9.83	$3.38	$28.09
Florida West (Gulf)	2446	$20.78	$10.10	$3.47	$28.87
Georgia	207	$20.23	$9.66	$3.40	$27.91
Louisiana	776	$20.67	$9.90	$3.38	$28.92
Mississippi	220	$20.85	$10.11	$3.48	$29.03
N. Carolina	603	$20.47	$10.00	$3.46	$28.62
S. Carolina	523	$20.89	$10.35	$3.60	$29.18
All States	6379	$20.62	$10.00	$3.44	$28.64

Table 2.4: Welfare Estimates from Southeast 1997 MRFSS-AMES Data

2.7.5 Benefit Transfer Welfare Estimation: Function Transfer

Since we have original welfare estimation results of marine recreational fishing value from the Northeast 1994 (Table 2.3) and Southeast 1997 data (Table 2.4) using the same benefit function (equation (2.13)), both regions could be a candidate for either the study site or the policy site for benefit transfer exercise. Function transfer procedure begins with inserting the policy site values into the independent variables of the study site benefit function. Using the study site benefit function and its parameter estimates with the policy site independent variable values, benefit transfer estimates of the economic value of marine recreational fishing for the policy site can be described as

$$(2.14) \quad WPT_{Study|Policy} = WTP_{BT} = (V^0_{Study|Policy} - V^1_{Study|Policy})/\beta_{1,Study}$$

where $WPT_{Study|Policy}$ is benefit function transfer welfare estimates for the policy site, $V^0_{Study|Policy}$ ($V^1_{Study|Policy}$) is the study site expected maximum utility function under a current (changed) policy regime adapted to the policy site context by inserting the policy site values into this study site benefit function's independent variables, and $\beta_{1,Study}$ is the study site parameter estimate of travel cost variable.

Table 2.5 (Table 2.6) shows benefit transfer welfare estimates for NE 1994 (SE 1997) using the benefit function and its parameter estimates from SE 1997 (NE 1994) model, $WTP_{SE97|NE94}$ ($WTP_{NE94|SE97}$). The benefit transfer estimates of the mean value of access per trip by state and wave show somewhat similar patterns with the policy site's original value estimates (Tables 2.3 and 2.4) in terms of the states with the largest

(Virginia and Florida for NE and SE respectively) and the lowest (New Hampshire and Alabama for NE and SE respectively) access value for both Northeastern and Southeastern coastal regions. As with the policy site's original estimates of access value per trip, benefit transfer estimates of access value do not show any clear pattern across waves in both Northeastern and Southeastern states. However, the benefit transfer estimates of marginal willingness to pay for historic harvest rate by species and state seem to carry the pattern that appeared in the study site's original estimation results in terms of the species group with the largest (flat fish and big game species groups for NE and SE respectively) marginal willingness to pay for a one fish increase in historic harvest rate for both Northeastern and Southeastern regions.

The Mean Value of Access Per Trip

State	All Waves	Wave 3	Wave 4	Wave 5	Wave 6
Connecticut	$5.89	$6.57	$6.15	$5.55	$4.76
Delaware	$3.40	$3.88	$3.33	$3.08	$3.35
Maine	$8.01	$9.52	$9.52	$8.98	$0.34
Maryland	$11.33	$12.32	$10.90	$12.10	$9.34
Massachusetts	$12.18	$12.32	$13.86	$13.06	$6.55
New Hampshire	$1.49	$1.71	$1.91	$1.45	$0.24
New Jersey	$15.24	$17.58	$15.22	$14.30	$13.03
New York	$24.04	$23.93	$23.55	$23.54	$26.22
Rhode Island	$8.29	$7.99	$9.22	$8.84	$5.71
Virginia	$40.16	$26.57	$31.97	$40.57	$80.90
Obs.	4897	1220	1675	1271	731

Willingness to Pay for a One Fish Increase in Historic Harvest Rate Per Trip

State	Obs.	Big Game	Small Game	Bottom Fish	Flat Fish
Connecticut	281	$1.76	$0.81	$0.18	$3.24
Delaware	190	$2.09	$0.93	$0.18	$3.86
Maine	273	$2.20	$1.05	$0.25	$4.80
Maryland	501	$2.29	$0.98	$0.22	$4.40
Massachusetts	529	$1.75	$0.77	$0.15	$3.26
New Hampshire	225	$1.98	$0.87	$0.18	$3.87
New Jersey	793	$1.79	$0.80	$0.14	$3.08
New York	678	$1.50	$0.66	$0.12	$2.57
Rhode Island	349	$1.67	$0.77	$0.16	$3.24
Virginia	1078	$2.13	$0.98	$0.17	$3.82
All States	4897	$1.90	$0.86	$0.17	$3.51

Table 2.5: Benefit Transfer Welfare Estimates for Northeast 1994 Using Southeast 1997 NRUM Parameter Estimates

The Mean Value of Access Per Trip

State	All Waves	Wave 2	Wave 3	Wave 4	Wave 5	Wave 6
Florida (All)	$205.70	$240.70	$196.95	$206.00	$185.34	$209.90
Florida West (Gulf)	$38.39	$47.23	$36.68	$35.68	$36.88	$37.28
Florida East (SA)	$10.49	$10.91	$8.82	$10.80	$10.06	$12.31
Georgia	$2.19	$0.75	$3.28	$2.86	$2.15	$1.52
N. Carolina	$25.44	$15.00	$26.80	$26.04	$34.09	$22.20
S. Carolina	$6.37	$6.41	$5.17	$5.99	$7.78	$6.52
Louisiana	$10.88	$7.89	$11.13	$12.75	$10.65	$11.63
Mississippi	$3.04	$2.87	$2.89	$2.90	$3.39	$3.09
Alabama	$1.34	$1.52	$1.63	$1.19	$1.02	$1.32
Gulf Coast	$74.07	$76.76	$71.33	$74.80	$75.08	$73.38
S. Atlantic	$113.29	$77.69	$117.13	$112.56	$140.84	$107.81
Obs.	6379	1039	1520	1115	1417	1288

Willingness to Pay for a One Fish Increase in Historic Harvest Rate Per Trip

State	Obs.	Big Game	Small Game	Bottom Fish	Flat Fish
Alabama	206	$38.39	$16.86	$17.04	$26.02
Florida East (SA)	1398	$38.67	$16.78	$17.66	$26.49
Florida West (Gulf)	2446	$38.67	$16.84	$17.68	$26.68
Georgia	207	$39.83	$16.87	$18.72	$26.98
Louisiana	776	$40.40	$17.31	$17.92	$28.00
Mississippi	220	$40.32	$17.62	$18.63	$27.82
N. Carolina	603	$40.55	$17.90	$19.31	$28.03
S. Carolina	523	$40.53	$18.27	$20.07	$28.00
All States	6379	$39.30	$17.13	$18.10	$27.06

Table 2.6: Benefit Transfer Welfare Estimates for Southeast 1997 Using Northeast 1994 NRUM Parameter Estimates

One way of empirically testing the validity of benefit transfer procedure is to compare benefit transfer welfare estimates for the policy site with the original welfare estimates available at the policy site (convergent validity test). The measure of convergent validity used in the analysis is

$$(2.15)\quad \delta_{BT} = (WTP_{BT} - WTP_{Policy}) / WTP_{Policy}$$

where δ_{BT} is the benefit transfer error measured as the percentage difference between benefit transfer estimates and the policy site's original estimates, WTP_{BT} is the benefit transfer welfare estimates for the policy site, and WTP_{Policy} is the original welfare estimates available at the policy site.

Tables 2.7 and 2.8 demonstrate the results of *convergent validity tests* of the benefit transfer welfare estimates for NE and SE regions respectively as described in equation (2.15). In the application of benefit function transfer procedure in a marine recreational fishing environment, the magnitude of benefit transfer error falls within 100% of the policy site's original welfare estimates in general except for the benefit transfer estimates of marginal willingness to pay for a one bottom fish increase in historic harvest rate for SE 1997 (above 400%). Benefit function transfer seems to perform better in estimating the mean access value of fishing sites than in estimating marginal willingness to pay for fishing quality in both regions with an exception of benefit transfer estimation of marginal willingness to pay for a one flat fish increase in historic harvest rate for SE 1997 (less than 8% of benefit transfer error). Another noticeable pattern is that benefit transfer estimates in both regions are generally underestimated compared to the policy site's original estimates except for the marginal willingness to pay estimates for a one fish increase in big game, small game, and bottom fish species groups for SE 1997.

The Mean Value of Access Per Trip

State	All Waves	Wave 3	Wave 4	Wave 5	Wave 6
Connecticut	11.03%	18.11%	7.96%	11.74%	4.12%
Delaware	40.64%	13.20%	19.69%	230.29%	33.61%
Maine	-57.31%	-53.08%	-59.50%	-58.86%	NA
Maryland	-61.80%	-62.51%	-61.07%	-66.34%	-45.80%
Massachusetts	-42.21%	-44.77%	-31.96%	-48.77%	-49.37%
New Hampshire	13.72%	-10.54%	25.66%	19.69%	NA
New Jersey	-56.33%	-57.02%	-54.15%	-58.96%	-54.92%
New York	-59.21%	-59.01%	-58.04%	-59.31%	-61.54%
Rhode Island	-16.38%	-12.26%	-10.88%	-20.52%	-30.17%
Virginia	-65.81%	-66.74%	-66.45%	-64.11%	-66.10%

Willingness to Pay for a One Fish Increase in Historic Harvest Rate Per Trip

State	Big Game	Small Game	Bottom Fish	Flat Fish
Connecticut	-91.96%	-89.97%	-97.03%	-79.90%
Delaware	-89.58%	-87.42%	-96.65%	-74.60%
Maine	-91.23%	-88.98%	-96.43%	-77.76%
Maryland	-91.06%	-89.53%	-96.66%	-78.52%
Massachusetts	-92.16%	-90.09%	-97.29%	-79.65%
New Hampshire	-91.35%	-89.18%	-96.94%	-77.63%
New Jersey	-90.15%	-87.76%	-97.13%	-76.27%
New York	-91.52%	-88.60%	-97.25%	-78.56%
Rhode Island	-91.93%	-89.79%	-97.03%	-79.37%
Virginia	-86.90%	-82.84%	-96.41%	-68.31%
All States	-90.46%	-87.94%	-96.87%	-76.40%

Table 2.7: Convergent Validity (Percentage Difference) Test of Benefit Transfer Estimates for Northeast 1994

The Mean Value of Access Per Trip

State	All Waves	Wave 2	Wave 3	Wave 4	Wave 5	Wave 6
Florida (All)	-31.46%	-31.53%	-31.50%	-31.31%	-31.49%	-31.46%
Florida West (Gulf)	-36.71%	-36.62%	-36.86%	-36.55%	-36.58%	-36.92%
Florida East (SA)	-35.80%	-35.86%	-36.12%	-35.89%	-35.15%	-36.00%
Georgia	-35.86%	-36.20%	-35.72%	-35.75%	-35.63%	-36.62%
N. Carolina	-31.60%	-31.02%	-32.33%	-31.50%	-31.24%	-31.57%
S. Carolina	-35.86%	-36.01%	-36.02%	-36.06%	-35.82%	-35.49%
Louisiana	-34.38%	-35.55%	-33.80%	-34.07%	-35.09%	-33.94%
Mississippi	-37.64%	-37.88%	-37.84%	-37.47%	-37.37%	-37.71%
Alabama	-35.98%	-35.91%	-35.70%	-35.75%	-36.34%	-36.31%
Gulf Coast	-34.69%	-35.28%	-34.65%	-34.35%	-34.61%	-34.64%
S. Atlantic	-30.23%	-30.70%	-30.31%	-30.34%	-29.96%	-30.12%

Willingness to Pay for a One Fish Increase in Historic Harvest Rate Per Trip

State	Big Game	Small Game	Bottom Fish	Flat Fish
Alabama	90.30%	72.17%	413.15%	-6.36%
Florida East (SA)	89.95%	70.63%	421.61%	-5.67%
Florida West (Gulf)	86.06%	66.68%	409.95%	-7.58%
Georgia	96.89%	74.63%	450.53%	-3.36%
Louisiana	95.44%	74.88%	429.90%	-3.15%
Mississippi	93.40%	74.30%	435.66%	-4.18%
N. Carolina	98.03%	78.99%	457.40%	-2.05%
S. Carolina	94.04%	76.45%	457.18%	-4.05%
All States	90.57%	71.20%	425.71%	-5.52%

Table 2.8: Convergent Validity (Percentage Difference) Test of Benefit Transfer Estimates for Southeast 1997

2.8 Conclusions

With two highly similar original valuation studies, the technique of *benefit function transfer* is applied to the valuation of marine recreation fishing in the coastal states of the Northeastern and Southeastern regions of the United States. Two welfare measures are estimated by transferring a two-stage nested random utility model of marine recreational fishing behavior: the *mean access value* per trip by state and wave and *willingness to pay for a one fish increase in five-year historic harvest rate* per trip by state and species group. The *convergent validity* of benefit function transfer procedure in a marine recreational fishing environment is empirically evaluated by examining the percentage difference between original and benefit transfer welfare estimates for the Northeast (1994) and Southeast (1997) coastal regions. Percentage differences between original and benefit transfer estimates for most benefit function transfer results are less than 100% of original welfare estimates. Benefit transfer estimation of site access value generally involves with smaller benefit transfer error than benefit transfer estimation of marginal willingness to pay for historic harvest rate of species groups. Benefit transfer estimates of marine recreational fishing value (site access value and marginal willingness to pay for historic harvest rate) for the Northeast and Southeast coastal regions seem to underestimate in general compared to original welfare estimates available at the same region.

One critical limitation of testing benefit transfer procedure with the NE 1994 and SE 1997 data is that the source of benefit transfer error cannot be clearly distinguished between regional and temporal variations. For function transfer, a behavioral relationship between marine recreational fishing and socio-economic and site characteristics variables is assumed to be identical at the policy and study sites. If this assumption doesn't hold because of regional (NE and SE) and/or temporal (1994 and 1997) variations, current data don't allow us to identify which variation is more responsible for benefit transfer error. Even when benefit transfer procedure adapts reasonably well to the differences in population and site characteristics, we still have two undistinguishable sources of benefit transfer error: regional and temporal variations that may lead to different behavioral relationships across different regions and points in time.

To identify which variation is more responsible for benefit transfer error, split sample intra-regional (within either NE or SE region) benefit transfer could be carried out by transferring welfare estimates or benefit function from the sample consisting of all regional states except for the policy state. The convergent validity of this split sample intra-regional benefit transfer can be tested by comparing original welfare estimates for the policy state from the sample consisting of all regional states including this policy state with benefit transfer welfare estimates transferred from the sample consisting of all regional states except for this policy state. The results of this test reflect more of the robustness of benefit transfer procedures than regional and/or temporal variations although they may still reflect some structural differences across states in the same region.

A comprehensive survey of benefit transfer's historical background, methodologies, and procedures helps us answer a question of when, why, and how to use this highly empirical technique of obtaining economic benefits (or costs) in a number of circumstances where the results of past research in a similar context are available. As is the case with most estimation techniques, benefit transfer has potential advantages and limitations with some necessary conditions for successful application that generates economically meaningful results. For valid and reliable benefit transfer results, benefit transfer practitioners should carefully consider the strength and weakness of the technique, and apply it only in feasible circumstances with appropriate professional judgments.

ESSAY 3

RECREATIONAL FISHING VALUE ESTIMATION OF WATER QUALITY IMPROVEMENTS IN WESTERN OHIO USING BENEFIT TRANSFER

ABSTRACT

This essay presents methods for estimating the value of recreational fishing trips and water quality improvements in two watersheds supporting a warm freshwater recreational fishery, the Stillwater River Watershed and Maumee River Basin, in western Ohio using benefit transfer. With improved water quality, the value of river-based recreation activities should increase as well as the number of total trips taken by recreators. Findings are that annual recreational fishing benefits of water quality improvements are $2,255,616 ($2,759,225 or $3,966,716) and $6,236,853 ($5,395,609 or $7,171,617) with about $44 ($54 or $77) and $58 ($50 or $66) of annual per angler benefits using average value transfer (two function transfer) estimates in the Stillwater River Watershed and Maumee River Basin respectively. These estimates along with disaggregated results in terms of local stream segments and angler types could serve as an initial set of approximated recreational benefits of any local environmental policy involving water quality improvement in inland streams and rivers, at least in terms of recreational fishing.

3.1 Introduction

As state and federal programs for farm conservation expand, questions are intensifying about the potential benefits of these expenditures. While there are undoubtedly many potential on-farm and off-farm benefits of conservation programs, this study focuses on estimating off-site river recreational benefits. With improved water quality, the value of river-based recreational activities should increase as long as there is no change in the cost of taking the trips (i.e., *per trip value increases*). In addition, the number of total trips taken by recreators may increase due to an increase in recreational value (*total trip increase*). To accurately assess the aggregate recreational fishing value of water quality improvements, increases in both the value of an individual trip and the number of total trips caused by a shift in recreation demand curve should be taken into consideration simultaneously.

Unfortunately, while there have been numerous non-market valuation studies conducted on water quality benefits, there are difficulties applying these studies to estimate the benefits of conservation programs. On the one hand, non-market valuation studies are often quite specific for particular resources or regions. They cannot necessarily be easily adapted to other areas. On the other hand, national studies, such as Ribaudo (1986) and Feather et al (1999), provide information that can assist national policy makers deciding where to target federal resources, but the complexity of water quality suggests that substantially more local information is necessary for local cost benefit analysis. At all scales, large river basins to small watersheds, there is often substantial variation in water quality. While federal and state laws often require each stream segment to meet federal standards, meeting these standards is often attempted with voluntary measures and conservation subsidy programs. Given limited resources available for these subsidy programs, local resource managers must have benefit cost information which they can use to make decisions about resources they do have.

This paper presents methods for estimating the benefits of recreational water quality improvements in two watersheds in Ohio using benefit transfer. The benefit transfer techniques used in this study adapt economic benefit estimates from other studies to a different region to estimate the value of recreational fishing trips and water quality improvements. The specific stream segments addressed are the Stillwater River Watershed and Maumee River Basin located in western Ohio. Both are freshwater streams supporting a warm water recreational fishery and recreational boating. These two watersheds are further disaggregated into several local stream segments within the watersheds to provide regional results for larger watersheds and to help policy makers target their efforts more carefully. Disaggregated estimation results in terms of both local stream segments and angler types (boat and shore anglers) can assist local policy makers and resource managers more effectively and efficiently carry out benefit cost analysis of local conservation programs or other environmental improvement programs.

3.2 Benefit Transfer Procedures

To illustrate how a meaningful benefit transfer study could be conducted, Table 3.1 shows the data needed to carry out benefit transfer. We first need to have adequate information about the nature of environmental good and/or change of interest (i.e., recreation type), site characteristics (i.e., site quality and available substitute sites), and population characteristics (i.e., income, age, and education) for the policy site (the place where original estimates are transferred to) and study site (the place with available original research results). These data on the context of the policy and study sites should be identified and compared to decide whether economically meaningful benefit transfer is possible. Benefit transfer applications provide economically more meaningful results when the attributes of environmental commodity, the nature of environmental change, and the characteristics of sites and affected populations display an adequate level of similarities between the policy site and the study site.

In addition to these basic data on environmental commodity attributes, site characteristics, and population characteristics, we need economic benefit estimates for value transfer or benefit function to predict non-market value for function transfer from the study site(s). The selected study site benefit estimates are directly transferred to the policy site for value transfer, while the policy site benefit is predicted using the selected study site benefit function, its parameter estimates, and the policy site values for the included variables (i.e., travel cost and income) for function transfer. The selected study site benefit estimates or benefit function should be adapted to the differences between the policy and study sites before attempting to directly transfer or to predict non-market value respectively. To aggregate directly transferred (value transfer) or predicted (function transfer) benefit estimates, both the mean number of trips per recreationist and the total number of relevant population must be multiplied by per trip value estimates. The above information is not always available in the reporting of data and estimation results from primary studies since most original research was not conducted for the future purpose of transferring estimated economic benefits or costs to the policy site of similar context.

Study Site Information Needed	***Policy Site Information Needed***
The nature of environmental good and/or change being evaluated	The nature of environmental good and/or change being evaluated
Study site characteristics: site quality and available substitute sites	Policy site characteristics: site quality and available substitute sites
Characteristics of affected population: socioeconomic variables (income, age, education and etc.)	Characteristics of affected population: socioeconomic variables (income, age, education and etc.)
Recreational value estimates (value transfer) or benefit function (function transfer) to transfer	Mean values for explanatory variables of benefit function: travel cost, income, substitute information and etc. (function transfer)
	Total number of affected population
	Mean number of trips per recreationist

Table 3.1: Information Requirements for Benefit Transfer

The necessary steps proposed and resulting templates for conducting benefit transfer are illustrated in Figure 3.1 and Table 3.2 respectively. Although the steps in Figure 3.1 are based on general recreation context, resulting templates (Table 3.2) are based on specific recreation activity (stream fishing) and regions (the Stillwater River Watershed and the Maumee River Basin in Ohio). Each study region is disaggregated into local stream segments as an attempt to provide recreational value estimates that can be used in both regional and local context. These proposed steps and following templates for value transfer and function transfer procedures are explained and demonstrated in great detail through the two applications (the Stillwater River Watershed and Maumee River Basin) in the following sections.

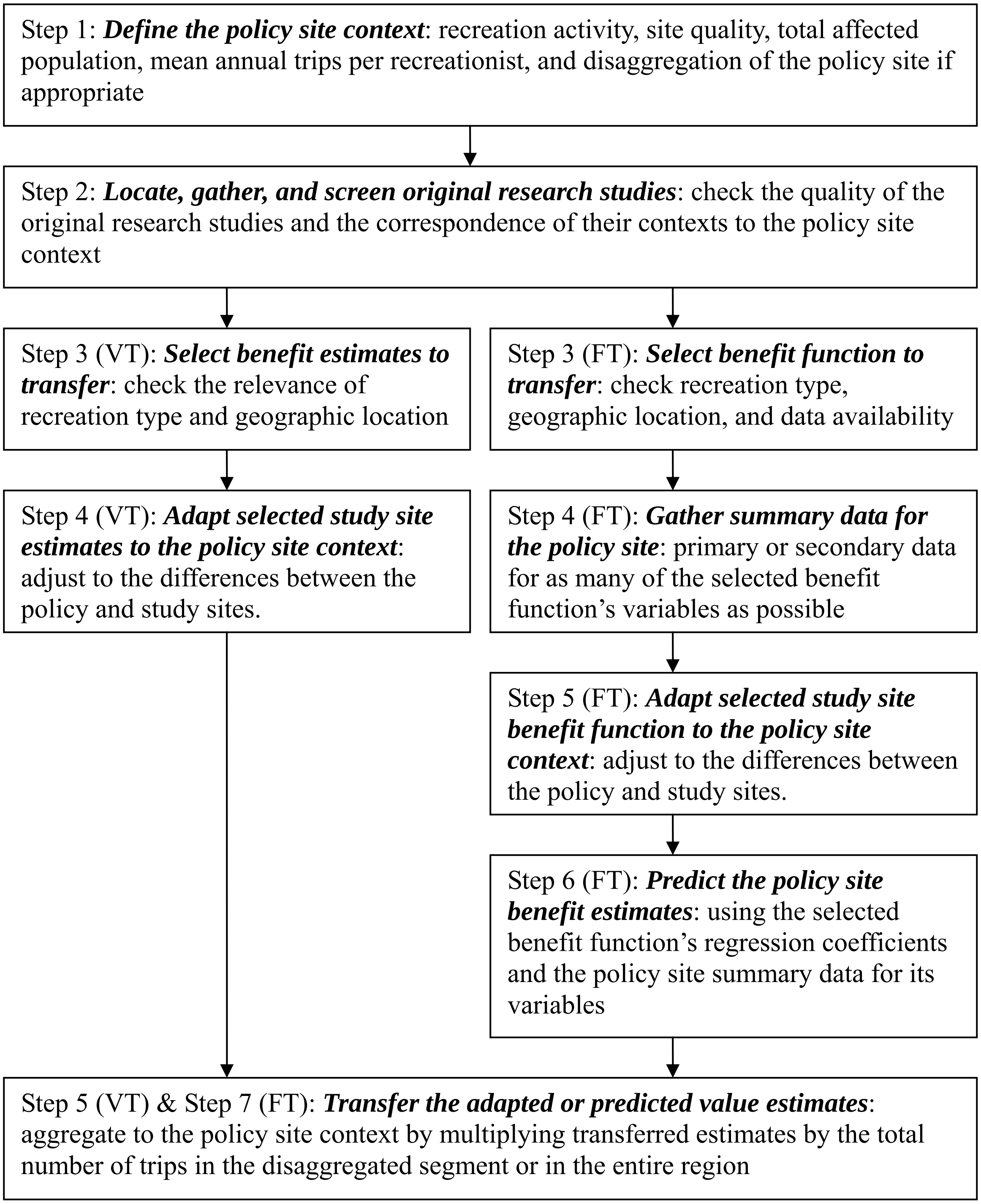

Figure 3.1: Steps for Value Transfer (VT) and Function Transfer (FT)

VT Steps	*Value Transfer Template*	*Function Transfer Template*	*FT Steps*
Step 1	N = Total number of anglers in the region	N = Total number of anglers in the region	Step 1
Step 1	X = Mean annual trips per angler in the region	X = Mean annual trips per angler in the region	Step 1
Step 1	P_i = The proportion of trips in the region allocated to the stream segment i	P_i = The proportion of trips in the region allocated to the stream segment i	Step 1
Steps 3 & 4 (VT)	CS_{Trip} = Transferred (Selected and adapted) value per trip	CS_{Trip} = Transferred (Selected, adapted, & predicted) value per trip	Steps 3, 4, 5, & 6 (FT)
Step 5 (VT)	$V_{i,Angler} = X*P_i*CS_{Trip}$ = Annual recreational value per angler for the segment	$V_{i,Angler} = X*P_i*CS_{Trip}$ = Annual recreational value per angler for the segment	Step 7 (FT)
Step 5 (VT)	$V_i = N* V_{i,Angler}$ = Total annual recreational fishing value for the segment	$V_i = N* V_{i,Angler}$ = Total annual recreational fishing value for the segment	Step 7 (FT)
Step 5 (VT)	$V_{Region} = \sum_i V_i$ = Total annual recreational fishing value for the region	$V_{Region} = \sum_i V_i$ = Total annual recreational fishing value for the region	Step 7 (FT)

Table 3.2: Templates for Benefit Transfer: Value Transfer and Function Transfer

3.3 Primary Studies of Freshwater Recreational Fishing Value

The question to be answered through the two benefit transfer applications of this study is "*What are the recreational fishing benefits associated with either maintaining excellent water quality condition or improving good, fair, or poor water quality conditions in the Stillwater River Watershed and Maumee River Basin located in western Ohio?*" Since we may not have enough resources available to conduct primary non-market valuation studies, one practical alternative of obtaining estimates of recreational fishing benefit would be to adaptively transfer available recreational fishing benefit estimates from another geographically or socio-economically similar regions (study sites) to the regions of interest without available benefit estimates (policy site). To carry out benefit transfer studies, the context of the policy site must be thoroughly defined, including recreation type and the characteristics of the site and relevant population, as illustrated in the step 1 of Figure 3.1 and Table 3.2. The Stillwater River Watershed and Maumee River Basin are two policy sites where we are trying to estimate recreational benefits of fresh, warm water fishing associated with either maintaining or improving water quality conditions in western Ohio's rivers and streams (step 1).

Once the policy site context is defined, relevant original research studies must be located, gathered, and screened as in the step 2 of Figure 3.1. Both the general quality (i.e., adequate data and sound methodology) and the specific correspondence to the policy site context (i.e., warm water, stream fishing of western Ohio's boat and shore anglers) of original non-market valuation studies should be examined. In Table 3.3, value estimates of a recreational fishing trip and per trip incremental value estimates due to water quality improvement obtained from various non-market valuation studies are presented. These benefit measures from original research studies are located and gathered through a thorough literature review, and screened for the general quality and the specific correspondence to the policy site context (step 2).

Although most benefit measures presented in Table 3.3 are original non-market valuation studies from various regions, Rosenberger and Loomis (2001) and Walsh, Johnson, and McKean (1992) are meta-analysis benefit transfer studies of outdoor recreation in the U.S. using benefit estimates of various outdoor recreation activities in 1967-1998 (41 recreational fishing benefit estimates) and 1968-1988 (23 warm water recreational fishing benefit estimates) respectively. The values from these two benefit transfer studies are summary measures of recreational fishing benefit estimates they use for their benefit transfer analysis, not meta-analysis regression function estimates (benefit transfer estimates). Summary measures from these two studies (Rosenberger and Loomis 2001; Walsh, Johnson, and McKean 1992) and value estimates from Bhat et al. (1998) are in terms of recreational fishing value per day; therefore, per day value measures are converted to per trip value measures by multiplying average days per warm water and cold water fishing trips in the U.S. (1.6 and 2) taken from Bergstrom and Cordell (1991). Recreational value estimates for cold water fishing may be converted to warm water fishing value by multiplying them by 0.56 since Bergstrom and Cordell (1991) show that the consumer surplus per trip for warm water fishing is about 56% of the consumer surplus per trip for cold water fishing in the U.S. Both original cold water fishing value estimates and converted warm water fishing value estimates from Bhat et al. (1998) are shown in Table 3.3.

As a screening process (step 2), gathered original studies are checked and compared for the level of the correspondence to the context of our policy sites. At least in terms of geographic location and relevant population, the results of Sommer (2001) may be the most relevant benefit estimates that could be adaptively transferred to our policy sites, the Stillwater River Watershed and Maumee River Basin. However, benefit estimates from Bergstrom and Cordell (1991), Walsh, Johnson, and McKean (1992), and Bhat et al. (1998) are also relevant because these studies provide *recreational warm water fishing* benefit estimates, i.e. the type of recreation activity to be analyzed in both of our policy sites. Further, there may be substantial differences in income levels among the group surveyed in Sommer (2001) and the relevant populations recreating at our policy sites. Sommer (2001), however, also provides estimates of the value of improved water quality condition that would be useful for our analysis.

For the purposes of this study, two methods of benefit transfer are conducted and compared. First, recreational fishing value of water quality improvements in the Stillwater River Watershed and Maumee River Basin are estimated by adaptively transferring actual recreational fishing value estimates (*value transfer*) presented in Table 3.3. Second, the demand functions estimated in these studies are adaptively transferred (*function transfer)* for two groups of anglers (boat and shore anglers) under two different water quality scenarios (current and maintained or improved water quality conditions).

Source	***Value per trip***	***Activity***	***Region/Population***
Value per Trip			
Hushak, Winslow, & Dutta (1988)	$8.18 - $17.07	Sport Fishing (Walleye)	*Lake Erie*
Bergstrom and Cordell (1991)	$27.71 - $30.07	*Warm Water Fishing*	U.S.
Walsh, Johnson, & McKean. (1992)	$42.34 - $65.26	*Warm Water Fishing*	
Englin, Lambert, & Shaw (1997)	$43.01 - $78.66	Lake Fishing	NY, NH, VT, & ME
Bhat et al. (1998)	$46.83 - $70.74 ($26.22 - $39.61)	Cold Water Fishing (*Warm Water Fishing*)	*Great Lakes States*
Rosenberger and Loomis (2001)	$27.56 - $47.85	Fishing	Northeast Region
McKean and Taylor (2001)	$28.05 - $45.37	Sport Fishing	Snake River, Idaho
Sommer (2001)	$11.36	Fishing	*Hocking River Valley, OH*
Value for Water Quality (W.Q.) Improvement			
Parsons and Kealy (1992)	$2.31 per trip	Increase DO[1] to 5 ppm[2]	Wisconsin
Feather et al. (1995)	$0.80 - $1.80 per trip	Improving water clarity	Minnesota Lakes
Englin, Lambert, & Shaw (1997)	$7.42 per trip	Increase DO to 5 mg/l[3]	NY, NH, VT, & ME
Sommer (2001)		*Improve water quality by 25%-50%*	*Hocking River Valley, OH*
	Per trip value: *25% imp: $7.70* *50% imp: $11.08* *Total trips:* *25% imp: +2.2* *50% imp: +4.5*		

1. DO = Dissolved Oxygen; 2. ppm = parts per million; 3. mg/l = milligram per liter

Table 3.3: Original Benefit Estimates of Recreational Fishing (1997 U.S. Dollar)

3.4 Data and Summary Statistics

As a first step for a benefit transfer study, the detailed context of the policy site must be defined (Figure 3.1 and Table 3.2). The data about Ohio's recreational boaters and anglers used to define the policy site context are primarily from three sources: the 1998 Ohio Recreational Boater Survey (Hushak 1999), 2001 Survey of Recreational Boater Safety and Participation in Ohio (Hushak 2002), and 2000 Fishing License Holder Information (the Ohio Department of Natural Resources: ODNR).

Hushak (1999) summarizes the results of the 1998 Ohio Recreational Boater Survey jointly supported by the Division of Watercraft, ODNR; the Boating Associations of Ohio/Lake Erie Marine Trade Association; the Lake Erie Protection Fund; and the Ohio Sea Grant College Program, the Ohio State University to evaluate an economic impact of recreational boating in the state of Ohio. Among approximately 239,816 registered boat-owning households (registered boats in Ohio/the mean number of boats owned) in 1998, a stratified random sample of 5,544 boat owners were surveyed with 2,339 respondents used in the analysis. The typical boat-owning household with a before-tax household income of $59,427 owned 1.7 boats with a length of 16-21 feet and a book value of $8,900 for the primary boat. The average number of annual boating trips to Ohio boating sites is 15.8 including trips to Lake Erie (4.3), Ohio River (1.3), inland lakes and reservoirs (8.7), and *inland rivers and streams* (1.5*)* with 38 miles of one-way trip distance, $134 per trip on trip-related expenditures, and the largest *time spent on fishing (50%)* followed by cruising (17%) and canoeing-kayaking-rowing (8%).

Hushak (2002) summarizes the results of the 2001 Survey of Recreational Boater Safety and Participation in Ohio jointly supported by the Division of Watercraft, ODNR; the Ohio Sea Grant College Program, the Ohio State University; and the National Oceanic and Atmospheric Administration (NOAA), U.S. Department of Commerce. Among approximately 219,581 registered boat-owning households in 2001, a random sample of 2,500 boat owners were surveyed with 692 respondents used in the analysis. The typical boat-owning household with a before-tax household income of $67,891 owned 1.9 boats with a mean length of 18.3 feet for the boat most used. One useful piece of information that is not included in the 1998 survey is *the mean number of people (2.6) on the boat* during a typical boating trip.

Since not all anglers use a boat while fishing and Sommer (2001) shows that only about a half of anglers use a boat to fish, it is also important to consider a group of anglers who do not use a boat when they fish, namely shore anglers. To estimate the numbers of potential boat and shore anglers at the policy sites, the total number of potential anglers, including both boat and shore anglers, is determined by using year 2000 resident fishing license holders' addresses with their zip codes available from the Division of Wildlife, ODNR.

3.5 Estimating the Number of Anglers

Once the type of recreation activity to evaluate (warm water stream fishing) is defined, the number of total relevant populations at our policy sites must be determined (step 1). Since most data (i.e., boat registration and fishing license holder data) used to identify relevant angler populations are or can be converted to county level data, the *study counties* to be included in our policy sites should be determined first. To approximate the total number of potential anglers in each watershed area, study counties are determined based primarily on their proximity to the public fishing or boating access points located within each watershed area. These counties are either located inside geographic boundaries (Figures 3.2 and 3.3) of watershed areas or located close enough to consider their residents as potential travelers to the public fishing or boating access points located within watershed areas if they are located outside geographic boundaries. Only counties in Ohio are considered in the analysis because the data about boat and shore anglers used to obtain benefit transfer estimates of recreational fishing value are available only in Ohio counties (Hushak 1999 and 2002; the ODNR 2000). As a result, six and twenty one Ohio counties are included in the Stillwater River Watershed and Maumee River Basin respectively. Detailed maps and addresses of Ohio's public boating facilities and public fishing waters are available from the websites of the ODNR's Division of Watercraft and Wildlife respectively (http://www.dnr.state.oh.us/watercraft/boat/opfg.htmand and http://www.dnr.state.oh.us/wildlife/Fishing/lakemaps/lmaps.htm).

Second, the *total angler population* in each study county in the analysis is estimated using zip codes from Ohio resident fishing license holders (the Division of Wildlife, ODNR), and by matching license holder zip codes to study counties in the Stillwater (six counties) and Maumee (twenty counties) regions. There are several zip codes available in each county, and one zip code sometimes belongs to more than one county. To avoid double counting of the people with zip codes belonging to more than one county, the number of Ohio resident fishing license holders with the same zip code is multiplied by the percentage of this zip code belonging to each study county. These weighted numbers of fishing license holders with same zip codes are then added together across all available zip codes in each study county to estimate the total number of potential anglers, resident fishing license holders, in the county.

Third, the *number of anglers who use a boat* when they fish in each study county is estimated based on the number of registered boats in 1998 and Hushak's 1998 and 2001 surveys on Ohio recreational boaters. To calculate the number of total boat-owning households in each county, the number of registered boats is divided by the mean number of boats owned by boat-owning households in the county. The number of total boat-owning households in the county is then multiplied by the mean number of people on the boat during a typical boating trip in Ohio for the year 2001 (2.6 people) to calculate the number of total boat recreators. To estimate the number of total boat anglers accounting for multiple recreational activities involved with a typical boating trip, the mean proportion of boating time spent on fishing during a boating trip in Ohio for the year 1998 (50%) is multiplied by the number of total boat recreators in each county.

Fourth, the *number of anglers who do not use a boat* when they fish is assumed to be the difference between estimated numbers of resident fishing license holders and boat anglers in each study county. Although some anglers may engage in both boat and shore fishing on the same trip, all anglers whose household has a registered boat are considered as boat anglers regardless of the possibility that they may engage in shore fishing. The number of shore anglers is likely to be underestimated because of the people on a boat during a typical boating trip without a fishing license, let alone shore anglers without a fishing license. For example, children under age 16 and senior residents over age 65 are not required to purchase a fishing license to fish in Ohio. Because current data do not allow us to distinguish between anglers with a fishing license and those without a fishing license, boat anglers without a fishing license are incorrectly subtracted and shore anglers without a fishing license are omitted from the total fishing license holder population in the calculation of shore anglers.

Finally, approximate numbers of *total boat and shore anglers* in the study counties of our policy sites can be calculated as

*(3.1) # of boat anglers = {# of registered boats/mean # of boats owned} * mean # of people on a boat * mean % of boat time spent on fishing*

(3.2) # of shore anglers = # of fishing license holders - # of boat anglers

where {# of registered boats (the ODNR)/mean # of boats owned (Hushak 1999) in each county} gives us the total number of boat-owning households in the county, mean # of people on a boat in Ohio is 2.6 (Hushak 2002), and mean % of boat time spent on fishing in Ohio is 50 % (Hushak 1999).

	Stillwater River Watershed	***Maumee River Basin***
Fishing License Holders	51,264	108,467
Boat-owning Households	21,509	38,528
Boat Recreators	55,923	100,173
Total Boat Anglers	*27,962*	*50,086*
Total Shore Anglers	*23,302*	*58,381*

Table 3.4: Estimation of Total Angler Population (Step 1)

Equations (3.1) and (3.2) provide approximate numbers of total boat and shore anglers in the study counties of the Stillwater and Maumee regions by combining available county data and some average values for Ohio recreational boaters taken from the ODNR and Hushak's surveys in1998 and 2001. Approximate numbers of boat and shore anglers in six and twenty one counties included in the Stillwater and Maumee regions respectively are added together to obtain total numbers of boat and shore anglers in each region. Final estimates of total boat and shore anglers at our two policy sites are presented in Table 3.4 along with some intermediate values: fishing license holders, boat-owning households, and boat recreators. The resulting proportions of boat anglers among total fishing license holders in both regions (55% and 46% in the Stillwater and Maumee regions respectively) seem to be consistent with Sommer (2001)'s results that 50 % of the anglers use a boat while fishing, and fishing is the primary activity of 59 % of the boaters for the Hocking River Valley in southeastern Ohio.

3.6 The Stillwater River Watershed

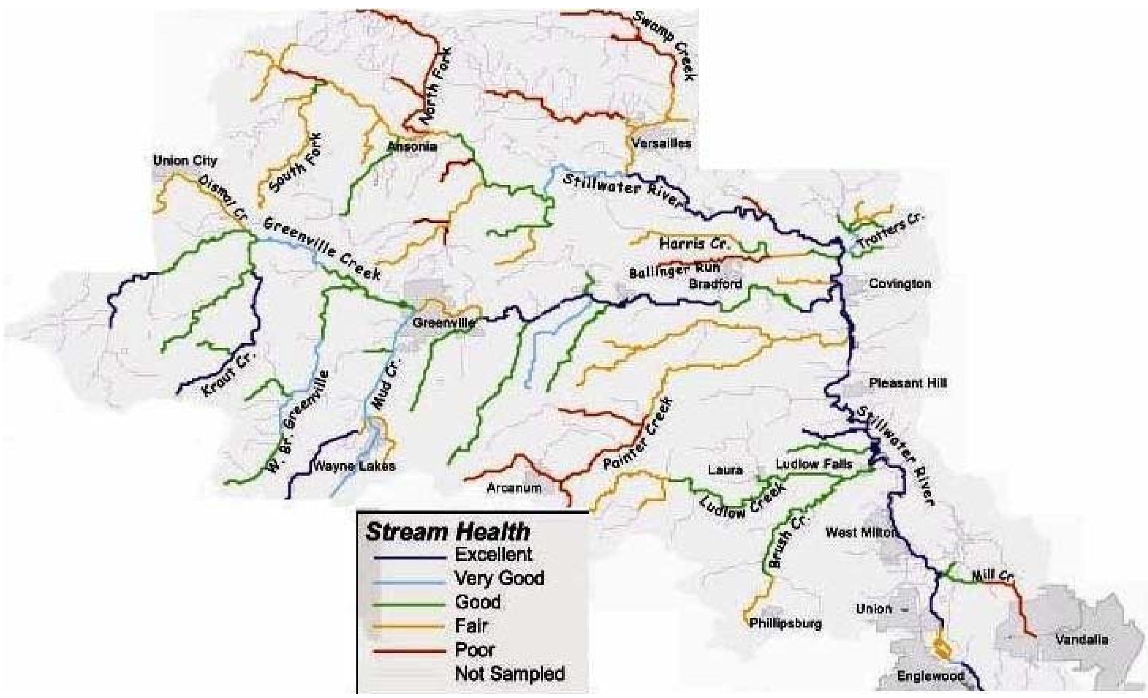

Figure 3.2: The Stillwater River Watershed
Source: Stillwater Watershed Group (data: Ohio EPA)

The drainage area and stream network with stream health conditions for the Stillwater River Watershed are shown in Figure 3.2. According to the Ohio Environmental Protection Agency (OEPA), the Stillwater River Watershed is impaired by nutrient enrichment, ammonia, metals, and other habitat alterations. Similar to many of the other watersheds in Ohio, the majority of the river miles are affected by nutrient enrichment. River and stream impairments commonly result from industrial wastes, human wastes, agriculture, and land use changes in the surrounding area. Despite impairments in many headwater areas, the mainstem of the Stillwater River and its major tributary, Greenville Creek, provide exceptional *smallmouth bass* fishing with excellent habitat and good water quality according to the ODNR.

As a first step for obtaining benefit transfer estimates of recreational fishing value at the policy site, the relevant angler population must be defined. According to Table 3.4, *total numbers of boat and shore anglers* (step 1) in the Stillwater River Watershed are 27,962 and 23,302 (N in Table 3.2) respectively. The mean number of *annual recreational fishing trips per angler* (step 1) to river streams is estimated by averaging annual boating trips to inland rivers and streams of six counties included in the Stillwater River Watershed region available from Hushak's 1998 survey on Ohio recreational boaters. The mean number of annual boating trips in the study counties is 2 (X in Table 3.2) for the Stillwater region, and it is applied both to boat anglers and to shore anglers assuming shore anglers take the same number of fishing trips per year as boat anglers. If shore anglers take more fishing trips than boat anglers possibly due to lower trip related expenditures, annual recreational fishing value estimates for shore anglers will be underestimated. Sommer (2001), however, shows that boaters (including fishers and non-fishers) and anglers (including boaters and non-boaters) take nearly the same number of trips (4.4 and 4.5 respectively) to the Hocking River Valley in Ohio.

For a more disaggregated and targeted analysis, the entire Stillwater region is divided into three stream segments based on HUC11 assessment unit classification (11 digit hydrologic unit code) by the Ohio EPA for Integrated Water Quality Monitoring and Assessment: the *Lower (or Mainstem) Stillwater River*, *Upper Stillwater River*, and *Greenville Creek* (step 1). Based on the availability of public fishing waters and boating facilities in each stream segment (the ODNR) supplemented with conversations with local anglers and fishing experts, total fishing trips of both boat and shore anglers in the entire region (annual fishing trips per angler (X) * total boat or shore anglers (N) in Table 3.2) are allocated to these three stream segments. One half of total annual fishing trips in the Stillwater region are allocated to the Mainstem Stillwater River segment with excellent water quality condition, and 30 % and 20 % of total fishing trips are allocated to the Upper Stillwater River segment with poor water quality condition and Greenville Creek segment with good water quality condition respectively (P_i in Table 3.2). Allocation of total fishing trips in the watershed to local stream segments is a critical assumption associated with regionalizing estimation results, i.e. estimating different recreation value for different regions within the watershed.

Water quality conditions (step 1) in each assessment unit are determined based on the number of stream site samples violating Ohio water quality standards established by the Ohio EPA using aquatic life use (e.g., pesticides, nutrients, turbidity, dissolved oxygen, siltation, unionized ammonia, pathogens, flow alteration, and other habitat alterations), recreation use (e.g., the secondary contact recreation maximum criteria of fecal coliform bacteria and E. coli bacteria), and fish consumption use (e.g., “Do Not Eat”, “One Meal per Week”, and “One Meal per Month” fish consumption advisories) assessments. The stream segment with “less than 25 %”, “25-50 %”, “50-75 %”, or “more than 75 %” of stream site samples violating Ohio water quality standards is assigned “excellent”, “good”, “fair”, or “poor” water quality condition respectively.

3.6.1 Value Transfer

The most important and also likely to be the most debatable part of value transfer process is to determine the *value of a typical recreational fishing trip* at the policy site with current and improved water quality conditions by adaptively transferring various benefit estimates available from the study site(s) (steps 3 and 4 in VT). Based on various recreational fishing value estimates from various geographic regions presented in Table 3.3, the range of *$12-$30* with an average transferred value of $20 ($CS_{Trip}$ in Table 3.2) is applied to the Stillwater region for the estimated baseline (with current water quality condition) value per fishing trip to both types of anglers. The range of $12-$30 for the value of a typical fishing trip in the Stillwater region is constructed based on the mean values from the studies conducted in geographically similar regions (Hushak, Winslow, and Dutta 1988; Sommer 2001) and the low bound values from other studies (Baht et al. 1998; Rosenberger and Loomis 2001; McKean and Taylor 2001) in Table 3.3 (step 3 in VT).

To account for the effect of water quality improvements on the total number of fishing trips (*total trip increase)*, maintaining excellent water quality condition and improving poor or good condition to excellent condition are assumed to increase the number of total recreational fishing trips by 50% in the entire Stillwater region (adjust X in Table 3.2). Water quality improvement is also assumed to increase the value of an individual fishing trip (*per trip value increase*) by a half of the average transferred value per fishing trip (50% of $20) if good water quality condition is improved to excellent condition or by the whole amount of average transferred value per fishing trip ($20) if poor condition is improved to excellent condition (step 4 in VT). The rationales behind these assumptions will be discussed in the next paragraph.

Assumptions on the magnitude of total trip and per trip value increases are based primarily on the results of Sommer (2001). Using both revealed and stated data, that study evaluates fishing and boating trips to the Hocking River Valley in southeastern Ohio providing estimates of changes in total trips and per trip value after water quality (dissolved oxygen, pH, biological criteria, and other pollutant concentrations obtained from the Ohio EPA) improvements. The study proposes two hypothetical water quality change scenarios: a small (25% increase in baseline streams meeting water quality standards) and a large (50% increase in baseline streams meeting water quality standards) improvement. Instead of taking actual values of changes in total trips and per trip value, the magnitude of proportional changes is adopted in the Stillwater region: 68% (98%) and 49% (100%) increases in per trip value and total trips respectively for a small (large) improvement in water quality. The estimation results for “small” and “large” water quality improvement scenarios from the Hocking River Valley in southeastern Ohio are adaptively transferred to estimate the economic value of "good to excellent" (50% increase in both total trips and per trip value) and "poor to excellent" (50% and 100% increases in total trips and per trip value respectively) water quality improvement scenarios for the Stillwater River Watershed in southwestern Ohio (step 4 in VT). If excellent water quality condition is maintained, no change is assumed in the value of an individual fishing trip to that segment although total fishing trips in that segment are assumed to increase because total fishing trips in the entire region are assumed to increase and the proportion of these trips allocated to each segment remains the same after water quality improvements (no change in P_i in Table 3.2).

Annual recreational fishing value per angler ($V_{i,Angler} = X*P_i*CS_{Trip}$ in Table 3.2) in each stream segment before and after water quality improvement are

(3.3) *BEFORE:*

Annual recreational value per angler with baseline water quality =

*(mean annual trips per angler to inland rivers/streams in the region) **

*(% of trips to the segment) **

(transferred value per fishing trip for the region)

(3.4) *<u>AFTER:</u>*

Annual recreational value per angler with water quality improvement =

*(mean annual trips per angler to inland rivers/streams in the region) **

*(1+ proportional change) **

*(% of trips to the segment) **

(original transferred value per trip for the region +

the value of water quality improvement for the segment).

For the results above, the mean annual trips per angler to inland rivers/streams in the Stillwater region is 2 (Hushak 1999); transferred values per trip for the region are $12, $20, or $30 (Hushak, Winslow, and Dutta 1988; Sommer 2001; Baht et al. 1998; Rosenberger and Loomis 2001; McKean and Taylor 2001); proportional change in the mean annual trips per angler for the region after water quality improvements is 50% (Sommer 2001); and the value of improvement for the segment is zero if excellent condition is maintained, 50% of the average transferred value per trip ($10) if good condition is improved to excellent condition, or 100% of the average transferred value per trip ($20) if poor condition is improved to excellent condition (Sommer 2001).

Equations (3.3) and (3.4) and Table 3.5 illustrate necessary steps to estimate annual recreational fishing value per angler in each segment before and after water quality improvement ($V_{i,Angler}$) along with important intermediate values used in these steps. By multiplying these per angler values obtained from Table 3.5 by the numbers of total boat and shore anglers in the region obtained during the first step (Table 3.4), we can estimate annual aggregate recreational fishing value in each segment (V_i = N* $V_{i,Angler}$ in Table 3.2) before and after water quality improvement (step 5 in VT). The difference between recreational fishing value before and after water quality improvement could be defined as the measure of the *recreational fishing value of water quality improvement* in each segment (step 5 in VT).

	Baseline Water Quality	***Improved Water Quality***
Mean Annual Trips in the Stillwater Region (Step 1)	2 (X)	3 (X*1.5)
% of Trips to the Segment (Step 1)	0.5, 0.3, or 0.2 (P_i)	0.5, 0.3, or 0.2 (P_i)
Transferred Value per Trip (Steps 3 & 4 in VT)	12, 20, or 30 (CS_{Trip})	CS_{Trip} + 0, 10, or 20
Annual Segment Value per Angler (Step 5 in VT)	$V_{i,Angler} = X*P_i*CS_{Trip}$	$V_{i,Angler} = (X*1.5)*P_i*(CS_{Trip} + 0, 10, \text{or } 20)$

Table 3.5: Annual Value per Angler for the Segment (Steps 3, 4, & 5 in VT)

Table 3.6 summarizes value transfer estimates of recreational fishing value in the Stillwater River Watershed (V_{Region} in Table 3.2) by each segment before and after water quality improvements for both boat and shore anglers. Either maintaining or improving to excellent water quality condition in the Stillwater region yields annual recreational fishing value of \$1,230,328 and \$1,025,288 on average for boat and shore anglers respectively adding up to \$2,255,616 for the entire Stillwater region (step 5 in VT). Annual per angler recreational fishing value of water quality improvement scenarios is \$44 for both types of anglers on average. With the lower bound transferred value per fishing trip, annual recreational fishing value of water quality improvements in the Stillwater region is \$1,845,504 with an annual per angler value of \$36.

Because twice as much of per trip value increase is assumed with "poor to excellent" water quality improvement scenario as with "good to excellent" scenario although total fishing trips are assumed to equally increase by 50% for the entire region with water quality improvements (Sommer 2001), the stream segment with poor baseline water quality may enjoy greater increase in recreational fishing value. It is also possible that the stream segment to which most trips are allocated realizes greater benefit from water quality improvement. In the Stillwater region, the largest increase in recreational fishing value comes from the *Upper Stillwater River* segment ($P_i = 0.3$) with poor baseline water quality condition (55% of total watershed value for both type of anglers) although most trips are allocated to the Mainstem Stillwater River segment ($P_i = 0.5$) with excellent baseline water quality condition. The effect of per trip value increase seems to be greater than the effect of total trip increase as a result of water quality improvements in the Stillwater region although the dominance of per trip value increase may be an artifact of assumptions. All regionalized results within the watershed in Table 3.6 depend on assumptions on baseline water quality condition, trip allocation (P_i), and changes in per trip value (CS_{Trip}) and total trips (X) after water quality improvement. For example, if the proportion of fishing trips (P_i) is assumed to increase or total fishing trips (X) is assumed to increase disproportionately more for the segment with greater water quality improvement, regional variation within the watershed will increase realizing even bigger increase in recreational fishing value for the Upper Stillwater River segment.

	Boat Anglers (N=27,962)			Shore Anglers (N=23,302)		
	Average ($20)	Lower ($12)	Upper ($30)	Average ($20)	Lower ($12)	Upper ($30)
Baseline Value						
Lower Stillwater	$559,240	$335,544	$838,860	$466,040	$279,624	$699,060
Upper Stillwater	$335,544	$201,326	$503,316	$279,624	$167,774	$419,436
Greenville Creek	$223,696	$134,218	$335,544	$186,416	$111,850	$279,624
Total Watershed	$1,118,480	$671,088	$1,677,720	$932,080	$559,248	$1,398,120
Improve W.Q.						
Lower Stillwater	$838,860	$503,316	$1,258,290	$699,060	$419,436	$1,048,590
Upper Stillwater	$1,006,632	$805,306	$1,258,290	$838,872	$671,098	$1,048,590
Greenville Creek	$503,316	$369,098	$671,088	$419,436	$307,586	$559,248
Total Watershed	$2,348,808	$1,677,720	$3,187,668	$1,957,368	$1,398,120	$2,656,428
Change in Value						
Lower Stillwater	$279,620	$167,772	$419,430	$233,020	$139,812	$349,530
Upper Stillwater	$671,088	$603,979	$754,974	$559,248	$503,323	$629,154
Greenville Creek	$279,620	$234,881	$335,544	$233,020	$195,737	$279,624
Total Watershed	***$1,230,328***	***$1,006,632***	***$1,509,948***	***$1,025,288***	***$838,872***	***$1,258,308***

Table 3.6: Value Transfer Estimates in the Stillwater River Watershed

3.6.2 Function Transfer

Instead of directly transferring estimates of recreational fishing value from other original studies, the recreation demand or benefit function(s) estimated at the study site(s) could be transferred to the policy site. Two versions of count-data travel cost recreation demand model are adaptively transferred to our policy sites: truncated Poisson model (Bhat et al. 1998) and Poisson random effect panel model (Sommer 2001) of recreational fishing trips. For simplicity and consistency, only simple Poisson model (equation (3.5)) estimation results of Sommer (2001) are considered along with truncated Poisson model (equation (3.6)) estimation results of Bhat et al. (1998) (step 3 in FT).

Two original recreation demand models considered are

$$(3.5) \quad \Pr(Y_i = y) = \exp(-\lambda_i)\, \lambda_i^{y} / y! \qquad y = 0,1,2,\ldots$$

$$(3.6) \quad \Pr(Y_i = y|Y_i > 0) = \exp(-\lambda_i)\lambda_i^{y}/y![1 - \exp(-\lambda_i)] \qquad y = 1,2,\ldots$$

where Y_i is the number of fishing trips taken by an angler i, $\Pr(Y_i = y)$ and $\Pr(Y_i = y|Y_i > 0)$ are probabilities of observing y trips from angler i, and λ_i is the conditional mean of a Poisson model. To estimate both simple and truncated Poisson models, λ_i can be parameterized as

$$(3.7) \quad E(Y_i \,|\, \mathbf{X}_i\boldsymbol{\beta}) = \lambda_i = \exp(\mathbf{X}_i\boldsymbol{\beta})$$

$$(3.8) \quad ln\lambda_i = \boldsymbol{X}_i\boldsymbol{\beta} + \varepsilon_i$$

where $\mathbf{X}_i$ is a vector of explanatory variables and $\boldsymbol{\beta}$ is a vector of coefficients. With this specification, a mean consumer surplus per trip is

$$(3.9) \quad \text{CS per trip} = 1/-\hat{\beta}_{TC}$$

where $\hat{\beta}_{TC}$ is the coefficient estimate of travel cost variable that is defined as a composite of variable costs of accessing a recreational site and the opportunity cost of time traveled.

Explanatory variables included in Bhat et al. (1998)'s demand function are household income, travel cost, travel cost to a substitute site, and a dummy for local participants, while Sommer (2001)'s demand function includes household income, travel cost, travel cost to a nearby recreation area, a dummy for fishing population, a dummy for the use of a powerboat, a powerboat dummy interacted with the boat's length, and trips to the outside of the Hocking River Valley. Common explanatory variables are household income, travel cost, and travel cost to a substitute site among which mean values for household income and travel cost for the Stillwater River Watershed could be estimated based on 2000 census and Hushak (1999) respectively (step 4 in FT). Since current data don't allow us to estimate travel cost to a logical substitute site, only household income and travel cost variables are included in the transferred demand function. Therefore, seemingly irrelevant variables to the policy sites and the variables of which mean values cannot be estimated due to data limitations are omitted from the transferred demand function (step 5 in FT).

Adaptively transferred demand function is

(3.10) $ln\lambda_i = \beta_0 + \beta_{TC}\ Travel\ Cost + \beta_{inc}\ Income + \varepsilon_i$

where λ_i is the number of fishing trips taken by an angler i and β_{TC} and β_{inc} are coefficients associated with travel cost and household income respectively. By inserting study site parameter estimates and available mean values for dependent and independent variables (Total Fishing Trips, Travel Cost, and Income) at the policy site into this transferred benefit function, benefit transfer estimates of recreational fishing value can be predicted using

(3.11) $\hat{\beta}_{\text{TC, Policy}} = (\ln\lambda_{\text{i,Policy}} - \hat{\beta}_{0\text{,Study}} - \hat{\beta}_{inc\text{,Study}}\ Income_{\text{Policy}}) / Travel\ Cost_{\text{Policy}}$

(3.12) Benefit transfer CS per trip = $1/-\hat{\beta}_{\text{TC, Policy}}$

where $\lambda_{\text{i,Policy}}$ is the mean total fishing trips at the policy site, $\hat{\beta}_{0\text{,Study}}$ and $\hat{\beta}_{inc\text{,Study}}$ are study site parameter estimates, $Income_{\text{Policy}}$ is the median household income at the policy site, and $Travel\ Cost_{\text{Policy}}$ is the mean travel cost at the policy site (step 6 in FT).

The mean annual fishing trips to river streams and median household income in the Stillwater region are 2 (Hushak 1999) and \$42,099 (2000 census) respectively. The mean annual fishing trips are assumed to increase by 50% (3) and 100% (4) with small (good to excellent) and large (poor to excellent) water quality improvements respectively (Sommer 2001). Mean travel cost is calculated as

(3.13) $\textit{Travel Cost}_{\text{Policy}}$ = Variable Costs (transportation and trip related expenditures) + Opportunity Cost of Time (time traveled for trip).

Variable costs of accessing fishing site include fuel cost for traveling (average round trip distance * federal employee mileage reimbursement rate in 2003 (\$.36)), fishing supply cost, boat launch fee, equipment rental, other boat trip supply cost, and boat fuel cost. Only a half of boat launch fee, equipment rental, other boat trip supply cost, and boat fuel cost are included in variable costs of accessing fishing site considering that only 50% of boating time during a typical boating trip in Ohio is spent on fishing activity (Hushak 1999). The opportunity cost of time traveled is approximated as 30% of foregone household income assuming 2080 work hours (52 weeks * 40 hours) to estimate hourly wage rate and average driving speed of 40 miles per hour to calculate travel time. The resulting mean travel cost (equation (3.13)) in the Stillwater region is \$64 per trip (step 4 in FT). Equations (3.11) and (3.12) along with these policy site values and study site parameter estimates provide, as shown in Table 3.7, function transfer estimates of recreation fishing value per trip under baseline and improved water quality scenarios (step 6 in FT).

	Bhat et al. (1998)	***Sommer (2001)***
Mean Annual Trips in the Stillwater Region (Step 4 in FT)	2, 3, or 4 ($\lambda_{i,Policy}$)	2, 3, or 4 ($\lambda_{i,Policy}$)
Intercept Parameter (Step 3 in FT)	2.8279 ($\hat{\beta}_{0\text{, Study}}$)	2.273 ($\hat{\beta}_{0\text{, Study}}$)
Income Parameter (Step 3 in FT)	-0.00000415 ($\hat{\beta}_{inc\text{,Study}}$)	0.000 ($\hat{\beta}_{inc\text{,Study}}$)
Median Household Income in the Stillwater Region (Step 4 in FT)	\$42,099 ($Income_{\text{Policy}}$)	\$42,099 ($Income_{\text{Policy}}$)
Mean Travel Cost in the Stillwater Region (Step 4 in FT)	\$63.98 ($Travel\ Cost_{\text{Policy}}$)	\$63.98 ($Travel\ Cost_{\text{Policy}}$)
Predicted Travel Cost Parameter in the Stillwater Region (Step 6 in FT)	-0.0306, -0.0243, or -0.0198 ($\hat{\beta}_{\text{TC, Policy}}$)	-0.0247, -0.0184, or -0.0139 ($\hat{\beta}_{\text{TC, Policy}}$)
Transferred Value per Trip (Step 6 in FT)	\$32.64, \$41.16, or \$50.5 (CS_{Trip})	\$40.5, \$54.48, or \$72.15 (CS_{Trip})

Table 3.7: Transferred Value per Trip (Steps 3, 4, 5, & 6 in FT)

These per trip value estimates derived from two study site demand functions (the last row of Table 3.7) are inserted into the value per trip (CS_{Trip}) before and after water quality improvement to obtain function transfer estimates of annual recreational fishing value for each stream segment (V_i in Table 3.2). Table 3.8 summarizes function transfer estimates of recreational fishing value in the Stillwater River Watershed (V_{Region} in Table 3.2) by each segment before and after water quality improvements for both boat and shore anglers. Function transfer estimates of annual aggregate recreational fishing value of water quality improvements in the Stillwater region are \$2,759,225 and \$3,966,716 with annual per angler value of \$54 and \$77 for both types of anglers by adaptively transferring recreational fishing demand functions from Bhat et al. (1998) and Sommer (2001) respectively (step 7 in FT). These function transfer estimates of recreational fishing value of water quality improvements are 22% (Bhat et al. 1998) and 76% (Sommer 2001) higher than average value transfer estimates.

As with value transfer, the largest increase in recreational fishing value comes from the *Upper Stillwater River* segment with poor baseline water quality condition (48% and 53% of total watershed value for both types of anglers using Baht et al. (1998) and Sommer (2001) respectively) showing more dominant effect of per trip value increase as a result of water quality improvements than the effect of total trip increase in the Stillwater region. Instead of simply assuming the magnitude of per trip value increases after water quality improvements as done in value transfer, per trip value increases are predicted by adapting study site demand functions to the policy site context (step 6 in FT) in function transfer although total trips are assumed to increase by 50% in all water quality improvement scenarios like value transfer. Therefore, the result that the Upper Stillwater River segment with poor current water quality captures the most benefit from water quality improvement in function transfer may reflect more systematic behavioral adjustment of local anglers due to the change in the quality of fishing sites, not an artifact of procedural assumptions.

	Boat Anglers (N=27,962)		***Shore Anglers (N=23,302)***	
	Bhat et al. (1998)	Sommer (2001)	Bhat et al. (1998)	Sommer (2001)
Baseline Value				
Lower Stillwater	\$912,741	\$1,132,389	\$760,628	\$943,671
Upper Stillwater	\$547,645	\$679,434	\$456,377	\$566,203
Greenville Creek	\$365,096	\$452,956	\$304,251	\$377,469
Total Watershed	\$1,825,482	\$2,264,779	\$1,521,257	\$1,887,343
Improve W.Q.				
Lower Stillwater	\$1,369,111	\$1,698,584	\$1,140,942	\$1,415,507
Upper Stillwater	\$1,270,911	\$1,815,831	\$1,059,107	\$1,513,214
Greenville Creek	\$690,481	\$914,013	\$575,409	\$761,688
Total Watershed	\$3,330,504	\$4,428,428	\$2,775,459	\$3,690,409
Change in Value				
Lower Stillwater	\$456,370	\$566,195	\$380,314	\$471,836
Upper Stillwater	\$723,266	\$1,136,397	\$602,731	\$947,011
Greenville Creek	\$325,385	\$461,057	\$271,158	\$384,220
Total Watershed	***\$1,505,022***	***\$2,163,649***	***\$1,254,203***	***\$1,803,067***

Table 3.8: Function Transfer Estimates in the Stillwater River Watershed

3.7 The Maumee River Basin

The Maumee River Basin drains 4.23 million acres from three states: Ohio, Michigan, and Indiana. Row-crop agricultural land (about 76%) and forest land comprise the majority of land use in the basin. The remainder of the region consists of urban and built-up land, and land devoted to rural transportation and various miscellaneous uses. The St. Joseph and St. Mary Rivers join at Fort Wayne, Indiana to form the Maumee River, which flows northeast to Lake Erie at Toledo, Ohio. The River has an average slope of 1.3 feet per mile. As shown in Figure 3.3, two other tributaries, the Tiffin River and the Auglaize-Blanchard River system, join the Maumee River at Defiance, Ohio. The Maumee River Basin is the largest tributary source of suspended sediment to Lake Erie with watershed erosion by water consisting of sheet rill erosion, gully erosion and stream bank erosion. Approximately 10.3 million tons of soil is detached from the soil surface in the Maumee River Basin on an annual average basis (Great Lakes Commission). The Maumee River, which attracts more than 60 % of fishing trips in the Maumee River Basin area, provides an excellent opportunity for *walleye* and *white bass* fishing (the ODNR).

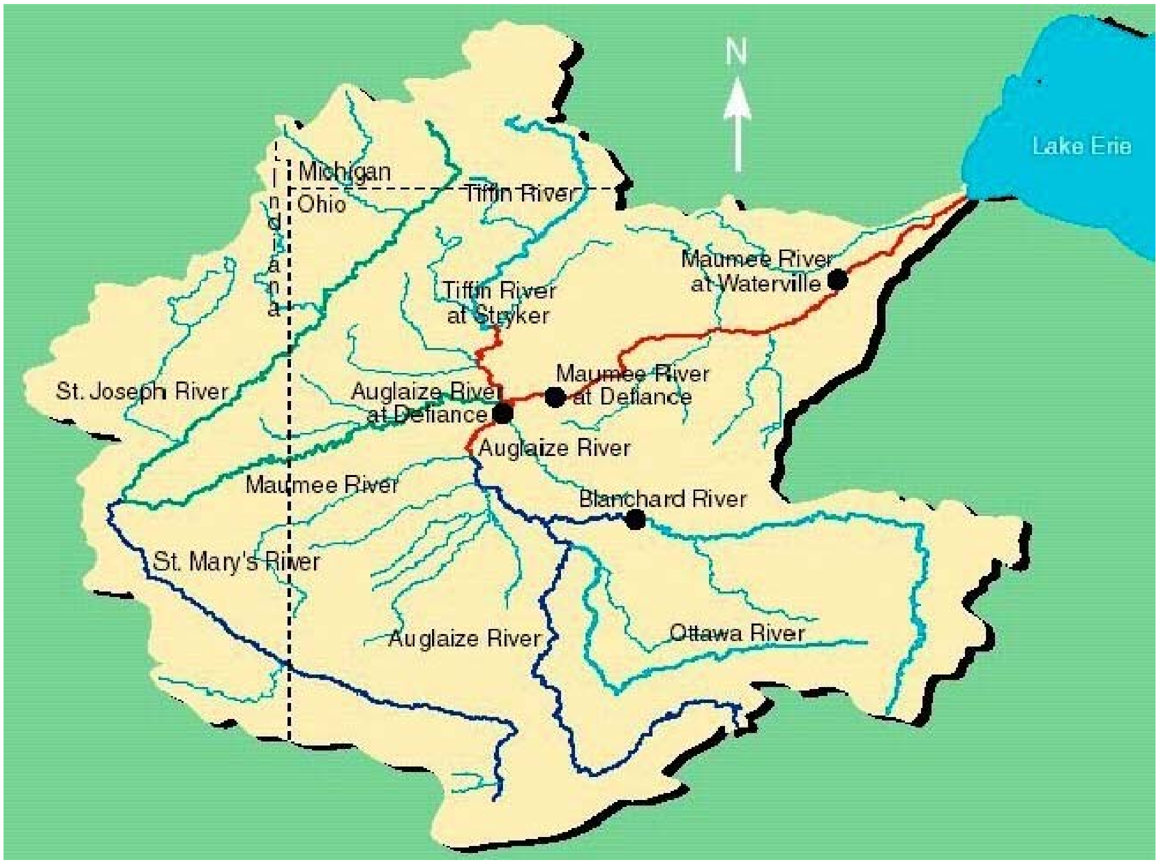

Figure 3.3: The Maumee River Basin
Source: Great Lakes Commission

According to Table 3.4, *total numbers of boat and shore anglers* (step 1) in the Maumee River Basin are approximated as 50,086 and 58,381 (N in Table 3.2) respectively that are more than twice as many anglers as in the Stillwater River Watershed. The mean number of *annual recreational fishing trips per angler* (step 1) to river streams is estimated by averaging annual boating trips to inland rivers and streams of 21 counties included in the Maumee River Basin region available from Hushak's 1998 survey on Ohio recreational boaters. As with the Stillwater region, the mean number of annual boating trips in study counties is 2 (X in Table 3.2) for the Maumee region, and it is applied to both boat anglers and shore anglers assuming shore anglers take the same number of fishing trips per year as boat anglers.

The Maumee region is also divided into seven stream segments for more disaggregated and targeted investigation according to the Ohio EPA's HUC11 assessment unit classification: the *Downstream Maumee River (from Defiance, Ohio to Lake Erie); Auglaize, Little Auglaize, Ottawa, and Blanchard Rivers; Upper Maumee River (from Defiance); Tiffin River; St. Joseph River; Toussaint River; and Portage River* (step 1). To allocate total recreational fishing trips in the Maumee region to each stream segment, more elaborate process is involved with greater use of the ODNR's data on public fishing waters and boating facilities in the Maumee River Basin area. All public fishing and boating access areas located near local stream segment (excluding fishing and boating access areas near lakes and reservoirs) are identified first, and the proportions of these local access areas in the entire Maumee region are calculated for public fishing and boating access areas separately by using the ODNR's detailed maps and address information about public fishing waters and boating facilities. Total fishing trips of both boat and shore anglers are allocated to seven local stream segments by averaging the proportions of local public fishing and boating access areas in the Maumee River Basin (P_i in Table 3.2). Most trips (60%) are allocated to the Downstream Maumee River followed by the Auglaize-Blanchard River system (18%), Portage River (9.5%), Toussaint River (7%), Upper Maumee River (3%), Tiffin River (2%), and St. Joseph River (1.5%).

Again, *water quality condition* (step 1) in each assessment unit is determined based on the number of stream site samples violating Ohio water quality standards established by the Ohio EPA using aquatic life use, recreation use, and fish consumption use assessments. Following the same rule used in the Stillwater region, water quality condition in each segment is determined by assigning "excellent" to the Downstream Maumee River, Auglaize-Blanchard River system, and St. Joseph River; "good" to the Upstream Maumee River, Tiffin River, and Portage River; and "poor" to Toussaint River.

3.7.1 Value Transfer

For the transferred *value of a typical recreational fishing trip* (steps 3 and 4 in VT) at the policy site, the range of \$30-\$50 with an average transferred value of \$40 (CS_{Trip} in Table 3.2) is applied to the Maumee region for the estimated baseline (with current water quality condition) value per fishing trip to both types of anglers. The range of \$30-\$50 for the value of a typical fishing trip in the Maumee region is constructed based on the ranges of values from the studies providing warm water fishing benefit (Bergstrom and Cordell 1991; Walsh, Johnson, and McKean 1992; Bhat et al. 1998) and the ranges of values from other studies providing more general fishing benefit (McKean and Taylor 2001; Rosenberger and Loomis 2001) in Table 3.3 (step 3 in VT). The applied range of transferred recreational fishing value estimates is higher than the range

($12-$30) applied to the Stillwater region assuming that recreational fishing value may differ depending on available species and stream type. Fishing in larger streams such as the Maumee River Basin with targeted species may provide anglers with larger recreation value than fishing in smaller streams such as the Stillwater River Watershed with less targeted species.

With the same assumptions used in the Stillwater case that adapts the results of Sommer (2001) to our policy site, maintaining excellent water quality condition and improving poor or good condition to excellent condition are assumed to increase the number of total recreational fishing trips (adjust X in Table 3.2) by 50% in the entire Maumee region (*total trip increase*). Water quality improvement is also assumed to increase the value of an individual fishing trip (*per trip value increase*) by a half of the average transferred value per trip (50% of $40) if good water quality condition is improved to excellent condition or by the whole amount of the average transferred value per trip ($40) if poor condition is improved to excellent condition (step 4 in VT). The estimation results for “small” and “large” water quality improvement scenarios from the Hocking River Valley in southeastern Ohio are adaptively transferred to estimate the value of “good to excellent” (50% increase in both total trips and per trip value) and “poor to excellent” (50% and 100% increases in total trips and per trip value respectively) water quality improvements for the Maumee Rive Basin in northwestern Ohio (step 4 in VT). If excellent water quality condition is maintained, no change is assumed in the value of an individual fishing trip although total trips are assumed to increase because total fishing trips in the entire region are assumed to increase and the proportion of these trips allocated to each segment remains the same after water quality improvements (no change in P_i in Table 3.2).

	Baseline Water Quality	***Improved Water Quality***
Mean Annual Trips in the Maumee Region (Step 1)	2 (X)	3 (X*1.5)
% of Trips to the Segment (Step 1)	0.6, 0.18, 0.95, 0.07, 0.03, 0.02, or 0.015 (P_i)	0.6, 0.18, 0.95, 0.07, 0.03, 0.02, or 0.015 (P_i)
Transferred Value per Trip (Steps 3 & 4 in VT)	30, 40, or 50 (CS_{Trip})	CS_{Trip} + 0, 20, or 40
Annual Segment Value per Angler (Step 5 in VT)	$V_{i,Angler} = X*P_i*CS_{Trip}$	$V_{i,Angler} = (X*1.5)*P_i*(CS_{Trip} + 0, 20, \text{or } 40)$

Table 3.9: Annual Value per Angler for the Segment (Steps 3, 4, & 5 in VT)

Equations (3.3) and (3.4) and Table 3.9 illustrate the process of estimating annual recreational fishing value per angler in each segment before and after water quality improvement ($V_{i,Angler}$) with intermediate values used in the process. For the results in Table 3.9, the mean annual trips per angler to inland rivers and streams is 2 (Hushak

1999); transferred values per trip are \$30, \$40, or \$50 (Bergstrom and Cordell 1991; Walsh, Johnson, and McKean 1992; Bhat et al. 1998; McKean and Taylor 2001; Rosenberger and Loomis 2001); proportional change in mean annual trips per angler for the region after water quality improvements is 50% (Sommer 2001); and the value of improvement is zero if excellent condition is maintained, 50% of the average transferred value per trip (\$20) if good condition is improved to excellent condition, or 100% of the average transferred value per trip (\$40) if poor condition is improved to excellent condition (Sommer 2001). Multiplying these per angler values from Table 3.9 by the numbers of total boat and shore anglers in the region from Table 3.4 provides annual aggregate recreational fishing value in each segment ($V_i = N^* V_{i,Angler}$ in Table 3.2) before and after water quality improvement (step 5 in VT). The difference between recreational fishing value before and after water quality improvement represents the *recreational fishing value of water quality improvement* in each segment (step 5 in VT).

Table 3.10 summarizes value transfer estimates of recreational fishing value in the Maumee River Basin (V_{Region} in Table 3.2) by each segment before and after water quality improvements for both boat and shore anglers. Either maintaining or improving to excellent water quality condition in the Maumee region yields annual recreational fishing value of \$2,879,945 and \$3,356,908 on average for boat and shore anglers respectively adding up to \$6,236,853 for the entire Maumee region (step 5 in VT). Annual per angler value of water quality improvement scenarios is \$58 for both types of anglers on average. With the lower bound transferred value per fishing trip, annual recreational fishing value of water quality improvements in the Maumee region is $5,141,335 with an annual per angler value of $47.

Unlike the Stillwater case where the largest value increase comes from the segment with poor baseline water quality, the largest increase in recreational fishing value comes from the *Downstream Maumee River* segment (P_i = 0.6) with excellent baseline water quality condition (42% of total watershed value for both types of anglers) followed by the Toussaint River segment (P_i = 0.07) with poor baseline current water quality condition. The effect of total trip increase as a result of water quality improvements seems to be more dominant than the effect of per trip value increase in the Maumee region. Again, all regionalized results within the watershed in Table 3.10 depend on assumptions on baseline water quality, trip allocation (P_i), and changes in per trip value (CS_{Trip}) and total trips (X) after water quality improvement. Therefore, estimation results for disaggregated stream segments should be interpreted with great caution, realizing that procedural assumptions may contribute significantly to important estimation results in a benefit transfer study especially.

	Boat Anglers (N=50,086)			Shore Anglers (N=58,381)		
	Average ($40)	Lower ($30)	Upper ($50)	Average ($40)	Lower ($30)	Upper ($50)
Baseline Value						
D. Maumee	$2,404,128	$1,803,096	$3,005,160	$2,802,288	$2,101,716	$3,502,860
Auglaize-Blanchard	$721,238	$540,929	$901,548	$840,686	$630,515	$1,050,858
U. Maumee	$120,206	$90,155	$150,258	$140,114	$105,086	$175,143
Tiffin	$80,138	$60,103	$100,172	$93,410	$70,057	$116,762
St. Joseph	$60,103	$45,077	$75,129	$70,057	$52,543	$87,572
Toussaint	$280,482	$210,361	$350,602	$326,934	$245,200	$408,667
Portage	$380,654	$285,490	$475,817	$443,696	$332,772	$554,620
Total Watershed	$4,046,949	$3,035,212	$5,058,686	$4,717,185	$3,537,889	$5,896,481
Improve W.Q.						
D. Maumee	$3,606,192	$2,704,644	$4,507,740	$4,203,432	$3,152,574	$5,254,290
Auglaize-Blanchard	$1,081,858	$811,393	$1,352,322	$1,261,030	$945,772	$1,576,287
U. Maumee	$270,464	$225,387	$315,542	$315,257	$262,715	$367,800
Tiffin	$180,310	$150,258	$210,361	$210,172	$175,143	$245,200
St. Joseph	$90,155	$67,616	$112,694	$105,086	$78,814	$131,357
Toussaint	$841,445	$736,264	$946,625	$980,801	$858,201	$1,103,401
Portage	$856,471	$713,726	$999,216	$998,315	$831,929	$1,164,701
Total Watershed	$6,926,894	$5,409,288	$8,444,500	$8,074,092	$6,305,148	$9,843,037
Change in Value						
D. Maumee	$1,202,064	$901,548	$1,502,580	$1,401,144	$1,050,858	$1,751,430
Auglaize-Blanchard	$360,619	$270,464	$450,774	$420,343	$315,257	$525,429
U. Maumee	$150,258	$135,232	$165,284	$175,143	$157,629	$192,657
Tiffin	$100,172	$90,155	$110,189	$116,762	$105,086	$128,438
St. Joseph	$30,052	$22,539	$37,565	$35,029	$26,271	$43,786
Toussaint	$560,963	$525,903	$596,023	$653,867	$613,001	$694,734
Portage	$475,817	$428,235	$523,399	$554,620	$499,158	$610,081
Total Watershed	***$2,879,945***	***$2,374,076***	***$3,385,814***	***$3,356,908***	***$2,767,259***	***$3,946,556***

Table 3.10: Value Transfer Estimates in the Maumee River Basin

3.7.2 Function Transfer

By inserting study site (Bhat et al. 1998; Sommer 2001) demand function's parameter estimates and available mean values for relevant variables at policy site into equations (3.11) and (3.12), function transfer estimates of recreation fishing value per trip (CS_{Trip}) could be predicted (step 6 in FT). The mean total annual fishing trips to river streams and median household income in the Maumee region are 2 (Hushak 1999) and \$41,433 (2000 census) respectively. As with the Stillwater region, the mean annual fishing trips are assumed to increase by 50% (3) and 100% (4) with small (good to excellent) and large (poor to excellent) water quality improvements respectively (Sommer 2001). Mean travel cost in the Maumee region is calculated with the same method used in the Stillwater region (equation (3.13)). The resulting mean travel cost in the Maumee region is \$79 per trip that is 23% greater than mean travel cost in the Stillwater region. Equations (3.11) and (3.12) along with these policy site values and study site parameter estimates provide, as shown in Table 3.11, function transfer estimates of recreation fishing value per trip under baseline and improved water quality scenarios (step 6 in FT).

These per trip value estimates derived from two study site demand functions (the last row of Table 3.11) are inserted into the value per trip (CS_{Trip}) before and after water quality improvement to obtain function transfer estimates of annual recreational fishing value for each stream segment (V_i in Table 3.2). Table 3.12 summarizes function transfer estimates of recreational fishing value in the Maumee River Basin (V_{Region} in Table 3.2) by each segment before and after water quality improvements for both boat and shore anglers. Function transfer estimates of annual aggregate recreational fishing value of water quality improvements in the Maumee region are \$5,395,609 and \$7,171,617 with per angler value of \$50 and \$66 by adaptively transferring recreational fishing demand functions from Bhat et al. (1998) and Sommer (2001) respectively (step 7 in FT). These function transfer estimates of recreational fishing value of water quality improvements are 13% (Bhat et al. 1998) lower and 15% (Sommer 2001) higher than average value transfer estimates.

	Bhat et al. (1998)	***Sommer (2001)***
Mean Annual Trips in the Maumee Region (Step 4 in FT)	2, 3, or 4 ($\lambda_{i,Policy}$)	2, 3, or 4 ($\lambda_{i,Policy}$)
Intercept Parameter (Step 3 in FT)	2.8279 ($\hat{\beta}_{0\text{, Study}}$)	2.273 ($\hat{\beta}_{0\text{, Study}}$)
Income Parameter (Step 3 in FT)	-0.00000415 ($\hat{\beta}_{inc\text{,Study}}$)	0.000 ($\hat{\beta}_{inc\text{,Study}}$)
Median Household Income in the Maumee Region (Step 4 in FT)	\$41,433 ($Income_{\text{Policy}}$)	\$41,433 ($Income_{\text{Policy}}$)
Mean Travel Cost in the Maumee Region (Step 4 in FT)	\$78.87 ($Travel\ Cost_{\text{Policy}}$)	\$78.87 ($Travel\ Cost_{\text{Policy}}$)
Predicted Travel Cost Parameter in the Maumee Region (Step 6 in FT)	-0.0249, -0.0197, or -0.0161 ($\hat{\beta}_{\text{TC, Policy}}$)	-0.0200, -0.0149, or -0.0112 ($\hat{\beta}_{\text{TC, Policy}}$)
Transferred Value per Trip (Step 6 in FT)	\$40.18, \$50.65, or \$62.12 (CS_{Trip})	\$49.92, \$67.16, or \$88.95 (CS_{Trip})

Table 3.11: Transferred Value per Trip (Steps 3, 4, 5, & 6 in FT)

As with value transfer results, the largest increase in recreational fishing value comes from the *Downstream Maumee River* segment (P_i = 0.6) with excellent baseline water quality condition (48% and 45% of total watershed value for both types of anglers using Bhat et al. (1998) and Sommer (2001) respectively) followed by the Toussaint River segment (P_i = 0.07) with poor baseline current water quality condition showing more dominant effect of total trip increase as a result of water quality improvements than the effect of per trip value increase in the Maumee region. More systematically adjusting behavioral changes of local anglers after water quality improvement by transferring the entire demand function to the policy site may provide estimation results possibly less sensitive to assumptions on baseline water quality condition, trip allocation (P_i), and changes in per trip value (CS_{Trip}) and total trips (X) after water quality improvement.

	Boat Anglers (N=50,086)		***Shore Anglers (N=58,381)***	
	Bhat et al. (1998)	Sommer (2001)	Bhat et al. (1998)	Sommer (2001)
Baseline Value				
D. Maumee R.	$2,415,208	$3,000,652	$2,815,203	$3,497,605
Auglaize-Blanchard R.	$724,562	$900,196	$844,561	$1,049,282
U. Maumee R.	$120,760	$150,033	$140,760	$174,880
Tiffin R.	$80,507	$100,022	$93,840	$116,587
St. Joseph R.	$60,380	$75,016	$70,380	$87,440
Toussaint R.	$281,774	$350,076	$328,440	$408,054
Portage R.	$382,408	$475,103	$445,741	$553,788
Total Watershed	$4,065,600	$5,051,098	$4,738,925	$5,887,636
Improve W.Q.				
D. Maumee R.	$3,622,812	$4,500,978	$4,222,805	$5,246,408
Auglaize-Blanchard R.	$1,086,844	$1,350,293	$1,266,841	$1,573,922
U. Maumee R.	$228,302	$302,749	$266,112	$352,888
Tiffin R.	$152,201	$201,832	$177,408	$235,259
St. Joseph R.	$90,570	$112,524	$105,570	$131,160
Toussaint R.	$653,405	$935,601	$761,619	$1,090,551
Portage R.	$722,956	$958,704	$842,688	$1,117,480
Total Watershed	$6,557,091	$8,362,682	$7,643,044	$9,747,669
Change in Value				
D. Maumee R.	$1,207,604	$1,500,326	$1,407,602	$1,748,803
Auglaize-Blanchard R.	$362,281	$450,098	$422,280	$524,641
U. Maumee R.	$107,541	$152,716	$125,352	$178,008
Tiffin R.	$71,694	$101,811	$83,568	$118,672
St. Joseph R.	$30,190	$37,508	$35,190	$43,720
Toussaint R.	$371,631	$585,525	$433,179	$682,497
Portage R.	$340,548	$483,600	$396,948	$563,692
Total Watershed	***$2,491,490***	***$3,311,584***	***$2,904,119***	***$3,860,033***

Table 3.12: Function Transfer Estimates in the Maumee River Basin

3.8 Conclusions

To develop estimates of recreational fishing value of water quality improvements, benefit transfer (*value transfer* and *function transfer*) techniques are applied to two regions with warm freshwater fishing environment, the Stillwater River Watershed and Maumee River Basin in western Ohio. Value transfer technique adapts value estimates from other regions (study site) to our regions of interest (policy site) by multiplying consumer surplus per trip estimates from the study site(s) by the predicted number of total fishing trips at the policy site, assuming that the change in welfare for an average individual is equivalent. Function transfer technique adapts demand or benefit function estimated in other regions to our regions of interest by inserting available policy site values for the variables included in the estimated study site demand function.

With value transfer, estimated annual recreational fishing benefits to both boat and shore anglers from either maintaining or improving to excellent water quality condition are $2,255,616 and $6,236,853 with annual per angler benefits of about $44 and $58 on average in the Stillwater and Maumee regions respectively. If we take low bound estimates to be conservative for benefit estimates, estimated fishing benefits from water quality improvements are $1,845,504 and $5,141,335 with annual per angler benefits of about $36 and $47 in the Stillwater and Maumee regions respectively. The Upper Stillwater River (55% of total watershed value) in the Stillwater region and Downstream Maumee River (42% of total watershed value) in the Maumee region are two local stream segments from which the largest recreational fishing benefits could be obtained with water quality improvements using average transferred values.

With function transfer, estimated annual recreational fishing benefits of water quality improvements are $2,759,225 and $5,395,609 with annual per angler benefits of about $54 and $50 by adaptively transferring the demand function from Bhat et al. (1998) while they are 3,966,716 and $7,171,617 with annual per angler benefits of about $77 and $66 by transferring the demand function from Sommer (2001) in the Stillwater and Maumee regions respectively. As with value transfer, the Upper Stillwater River (48% and 53%) in the Stillwater region and the Downstream Maumee River (48% and 45%) in the Maumee region are two stream segments from which largest recreational fishing benefits could be obtained with water quality improvements (using Bhat et al. 1998 and Sommer 2001).

In comparison to average value transfer estimates, function transfer estimates are 22% greater and 13% smaller using Bhat et al. (1998) while they are 76% and 15% greater using Sommer (2001) in the Stillwater and Maumee regions respectively. These differences in aggregate benefit estimates seem to be caused primarily by different assumptions on the value per trip (CS_{Trip}) under different water quality scenarios in value transfer. The assumed values per trip for baseline, good to excellent, and poor to excellent water quality scenarios are $20, $30, and $40 for the Stillwater region and $40, $60, and $80 for the Maumee region respectively. Compared to the assumed per trip values in value transfer, the predicted per trip values in function transfer are 42% greater and 19% smaller using Bhat et al. (1998) while they are 88% and 16% greater using Sommer (2001) on average in the Stillwater and Maumee regions respectively. These percentage differences in the assumed (value transfer) and predicted (function transfer) per trip values display similar pattern with percentage differences in the value transfer and function transfer estimates.

There are several sources of measurement errors and possibilities of improvement associated with current benefit transfer estimates of recreational fishing value in the Stillwater and Maumee regions.

First, the use of information about boat anglers to make important behavioral assumptions on shore anglers may cause measurement errors. The mean annual trips may be different between boat and shore anglers although Sommer (2001) shows that the number of trips to the Hocking River Valley in southeastern Ohio are nearly the same among boaters (both anglers and non-anglers) and anglers (both boat and shore anglers). To correctly calculate the total number of recreational fishing trips, we need to know the mean annual fishing trips by shore anglers. In addition, a mean consumer surplus per trip could also differ between boat and shore anglers since a boating trip usually involves more than one recreation activity. A consumer surplus per boating trip may not even be possible to be logically divided into several activities involved in a typical boating trip.

Second, we assume that all the people on a boat are fishing license holders to calculate the number of shore anglers out of resident fishing license holder population. Resident fishing license is required for the anglers of age 16-65 who have resided in Ohio for the past six months. Since both boat and shore anglers would definitely include children and/or senior residents, the number of people on a boat without fishing license is incorrectly subtracted and the number of shore anglers without a fishing license is omitted in the calculation of the total number of shore anglers. As a result, benefit transfer estimates of water quality improvements for shore anglers will be underestimated assuming the transferred consumer surplus per fishing trip is correct.

Third, current recreational fishing benefit estimates from water quality improvements ignore the possibility of new addition to the current angler population, especially in the segment with poor baseline water quality condition. A stream segment with improved water quality could attract not only more fishing trips from exiting anglers but also fishing trips from new anglers who are not currently included in the angler population. By omitting potential trips from new anglers after water quality improvements, benefit transfer estimates of recreational fishing could be underestimated.

Fourth, our assumptions on proportional changes in the number of total fishing trips and the value of each fishing trip after water quality improvements in both value transfer and function transfer should be refined further by taking account of the characteristics of affected angler population (e.g., household income), site attributes (e.g., unique habitats), diminishing nature of marginal utilities from water quality improvements, and possibilities of nearby substitutes not included in our current analysis. In the process of constructing benefit transfer estimates of recreational fishing value, it is very important to adapt to the differences in relevant characteristics of population, site, and substitute. In addition, these assumptions could significantly affect our localized benefit measures especially for value transfer estimates.

Fifth, assumptions on the proportions of trips from each segment in each region are not adjusted to water quality improvement scenarios. It is very likely that water quality improvements could affect the proportional trips from each segment since different segments have different baseline water quality conditions. The stream segment with most improvement in water quality may attract more anglers from other segments, let alone additional trips from new and existing anglers in the same segment. The stream segment with no improvement may lose some trips to another segment with more improvement in water quality. Although aggregate benefit estimates may not be affected by changes in proportional trips within each region, benefit estimates from local stream segment should be carefully interpreted. Benefit estimates for the segment with most water quality improvement could be underestimated while benefit estimates for the segment with no water quality improvement could be overestimated under current assumption.

Finally, even with a heroic assumption that transferred benefit estimates per trip using *value transfer* are correct measure of recreational fishing benefit at our policy sites, we need a linear (linear with trip cost) demand curve for recreational fishing trip to minimize errors associated with assumptions on changes in number of total trips and value per trip after water quality improvements. Since a mean consumer surplus per trip is simply multiplied by the total number of potential trips at the policy site to approximate an aggregate benefit, nonlinearity in demand could introduce more sources of error in estimating an aggregate benefit measure. In general, *function transfer* technique can adjust for differences in site and population characteristics more systematically by directly transferring whole demand or benefit function from a study site; however, statistically similar benefit functions could still yield statistically different welfare measures when benefit measures are a nonlinear function of estimated coefficients (Downing and Ozuna 1996).

Although there are many different sources of measurement error, benefit transfer estimates of recreational fishing value presented here could serve as an initial set of approximate recreational benefit estimates of any environmental policy involving water quality improvements in inland streams and rivers, at least in terms of recreational fishing. To be conservative on the benefit side of cost-benefit analysis, it may be safer to take low bound estimates of recreational fishing value presented in this study. In fact, some sources of measurement error could lead us to underestimate benefit measures although some sources cannot be clearly determined if they could bias our benefit estimates upward or downward. As is always the case with benefit transfer process itself, careful professional judgments and efforts should be practiced before adopting benefit transfer estimates of any recreation activity at any stage of policy formulation.

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