Measuring the Economic Benefits of Water Quality Improvement with Benefit Transfer: An Introduction for Noneconomists

CHRISTOPHER F. DUMAS, PETER W. SCHUHMANN

Department of Economics and Finance, University of North Carolina at Wilmington
Wilmington, North Carolina 28403-5945, USA

JOHN C. WHITEHEAD*

Department of Economics, Appalachian State University, Boone, North Carolina 28608-2051, USA

Abstract.—In this paper, we provide an introduction to water quality benefit estimation for noneconomists. Net water quality benefits are typically measured using the concept of consumer surplus, which is estimated using a number of economic valuation methodologies. These are divided into direct and indirect methods. Direct methods involve questioning survey respondents to determine their consumer surplus. Indirect methods use data from consumer market behavior to estimate economic values. When limited time or funding preclude costly data collection and the development of new consumer surplus estimates, the method of benefit transfer is used to tailor preexisting consumer surplus estimates to fit new policy situations. We provide an example of benefit transfer by estimating the value of water quality improvements for the Cape Fear River in North Carolina. Benefit transfer methods are used with three valuation approaches to estimate the benefits of water quality improvement.

Introduction

Urbanization has negative impacts on river and stream water quality and associated economic benefits. This chapter will describe categories of water quality benefits, discuss the economic methodologies commonly used to estimate the values of these benefits, explore the relatively new techniques of benefit transfer used to estimate benefits of a given water quality improvement using information from other locations or time periods, and apply benefit transfer techniques in a case study of the benefits of water quality improvement in the Cape Fear River basin, North Carolina. The discussion illustrates how economic methodologies can be used to document the economic benefits of maintaining water quality and associated ecological functions.

Water quality provides two broad classes of economic benefits, withdrawal benefits, and instream benefits (Feenberg and Mills 1980). Withdrawal benefits include municipal water supply and domestic

use (e.g., household drinking, cooking, washing, and cleaning) benefits, agricultural irrigation and livestock watering benefits, and industry process water benefits. If water quality is low, withdrawn water must be treated before it can be used, and the economic benefits (net of treatment costs) associated with its use are lower. Instream benefits (i.e., the benefits of water quality arising from water left "in the stream" and not withdrawn) include two subcategories: use benefits and nonuse benefits. Instream use benefits include swimming, boating, and sport-fishing benefits—benefits associated with direct human interaction with water in the stream/river. Other instream use benefits include the esthetic value of water quality that may accrue to nearby picnickers, streamside trail hikers, and streamside property owners. Instream *nonuse* benefits of water quality include stewardship value, altruistic value, bequest value, and existence value. Nonuse benefits accrue to individuals regardless of whether or not they have direct interaction with water. Stewardship value arises from a belief (often moral or religious) that humans are responsible for maintaining some level of

* Corresponding author: whiteheadjc@appstate.edu

water quality even in cases where no withdrawal or instream use benefits result. Altruistic value arises from the enjoyment some people receive from simply knowing that other people enjoy withdrawal or instream use benefits. Bequest value arises from a belief that current human generations are responsible for maintaining some level of water quality to "bequest" to future human generations. Existence value arises from the enjoyment some people receive from simply knowing that some level of environmental quality exists. If water quality is allowed to deteriorate, then stewardship, bequest, and existence goals may not be met, and associated benefits fall.

The impacts of urbanization on water quality benefits are mediated by aquatic ecosystems. Increases in stream nutrient levels that lead to algae blooms can reduce swimming and boating benefits. Reductions in dissolved oxygen that lead to fish kills can reduce fishing and streamside property value benefits. Increases in disease-causing bacteria due to urban and suburban storm water runoff can increase water treatment costs and reduce swimming, fishing, and boating benefits. Reductions in aquatic species populations or diversity caused by stream sedimentation or toxic chemical discharges can reduce stewardship, altruistic, bequest, and existence values. Economic valuation methodologies typically trace changes in water quality variables through changes in aquatic ecosystem parameters to changes in economic benefits. Often, it is a change in an aquatic ecosystem parameter, such as a fish population, algae population, or disease-causing bacteria population, that is the ultimate cause of a change in economic benefits.

The economic valuation methodologies vary depending on the category of water quality benefit. Appropriate methodologies are used to estimate the benefits arising from each category, and the resulting benefits are then added to arrive at a measure of the overall value of water quality. Market prices can be used together with traditional economic valuation and benefit cost analysis methodologies to derive estimates of most withdrawal benefits. However, many instream benefits lack direct market prices and have public good characteristics that make benefit estimation using traditional economic methodologies difficult. Specifically, many instream benefits exhibit the public good characteristics of *nonrivalry* and *nonexcludability*. Nonrivalry means that more than one consumer can enjoy the quality of a given body of water at the same time (whether or not this enjoyment is associated with direct use). Nonexcludability means that it is difficult (costly) to prevent one individual from enjoying the benefits created by another individual's actions. If individuals cannot be excluded, then they will not pay prices to gain the benefits (instead, they will "free ride"), and therefore, price data will not be available.

This paper will focus on the estimation of the instream benefits of water quality changes because these types of benefits are more difficult to estimate and they are most pertinent to the theme of this book. Instream benefits are typically estimated using nonmarket valuation methodologies. Nonmarket techniques have been developed to estimate economic values in situations where direct market prices are lacking and where public good characteristics are significant. Nonmarket valuation methodologies include direct or stated preference and indirect or revealed preference approaches. The contingent valuation, contingent behavior, and conjoint/choice analysis methods are examples of direct approaches. The travel cost, averting behavior, and hedonic price methods are indirect approaches. Each of these methods requires primary data collection. When the cost of primary data collection is prohibitive and/or time is short, the benefit transfer approach can be used to develop economic benefit estimates.

With benefit transfer, benefit estimates from existing direct or indirect valuation case studies are spatially and/or temporally transferred to a new case study. There are four types of benefit transfer approaches: benefit estimate transfer, benefit function transfer, meta-analysis, and preference calibration. Benefit estimate transfer uses summary measures of the environmental benefit estimates directly. Researchers simply obtain a benefit estimate from a similar study conducted elsewhere and use it for the current policy analysis case study. With benefit function and meta-analysis transfer, researchers use statistical models to transfer benefits. Characteristics of the current policy situation or case study (e.g., population demographics, site characteristics) are substituted into a statistical model to translate benefit estimates more accurately. Preference calibration uses an analytical model to reconcile existing benefit estimates derived from different methodological approaches and develop consistent benefit estimates for the new policy study.

The remainder of this paper is organized as follows. In the second section of the paper, we present the economic theory and some definitions used in benefit–cost analysis and describe the water quality valuation methodologies. In the third section, we discuss the benefit transfer approach to estimating water quality benefits. In the fourth section, we present a case study: water quality improvement in the Cape

Fear River. In this example, benefit transfer methods are used with three valuation approaches to estimate the benefits of water quality improvement. The fifth section is a summary of our findings.

Economic Theory

Whenever a government project or policy is implemented, there are economic winners and losers. The economic efficiency criterion requires that the gains to the winners exceed the losses imposed on the losers. Economic efficiency is one of several criteria (others include equity and risk) used to assess the desirability of government projects, such as water quality improvement projects. Benefit–cost analysis is a method used to calculate and compare monetary gains and losses for the purpose of assessing efficiency (Boardman et al. 2001). When government pursues a water quality improvement policy, such as the regulation of polluting firms or the implementation of urban land use controls (e.g., zoning), gains and losses are distributed to consumers and firms. Losses are typically relatively straightforward to measure by considering reductions in firm profits and increases in consumer costs. However, gains are often more difficult to measure, especially when they come in the form of public goods such as water quality.

The concept of consumer surplus is the basis for measuring net economic benefits. Considering a market good, for example a car, the consumer surplus is the difference between what the consumer is willing (and able) to pay and the market price (amount actually spent) for the car. Consumer surplus is also called net willingness to pay (net WTP) since it is willingness to pay net of the costs. The consumer may be willing and able to pay the manufacturer's suggested retail price of $35,000 for a new Ford Mustang. However, if the agreed-upon price is $31,000 then the consumer surplus is $4,000—the difference between the consumer's maximum willingness to pay and the market price.

Nonmarket goods such as water quality also provide consumer surplus (Freeman 1993). In the context of water quality valuation, suppose a catch-and-release freshwater angler is willing and able to pay up to $125 for a good day of urban fishing. If the cost of the day trip is $25, then his consumer surplus is $125 – $25 = $100. Now suppose that a zoning law is enacted that leads to a water quality improvement that, in turn, increases the angler's expected catch per trip. With the increase in expected catch, the angler's willingness to pay might increase to, say, $160. If so,

the angler's consumer surplus per trip after the water quality improvement is $160 – $25 = $135. The angler's economic gain from the water quality improvement is the change is his or her consumer surplus, or $135 – $100 = $35. The empirical challenge, of course, is to determine the angler's willingness to pay and consumer surplus before and after the water quality change.

Economics students may remember the graphical depiction of demand and consumer surplus (Figure 1). The demand curve (denoted D_1) is a downward sloping line with market price on the vertical axis and quantity purchased/consumed on the horizontal axis. The demand curve slopes downward due to the fact that lower prices are required to convince consumers to purchase larger quantities. Typically, the position of the demand curve is estimated using data on market prices and quantities purchased by the consumer. The rectangle below the current market price is the initial expenditure on the good (i.e., the product of price per unit and quantity of units purchased, noted as EXP in Figure 1). Changes in consumer surplus and not changes in expenditures (DEXP) should be used in benefit–cost analysis (Edwards 1991). In Figure 1, consumer surplus (CS) is the triangular area above the current market price and below the demand curve. The area of the consumer surplus triangle increases or decreases with changes in demand (i.e., with shifts in the position of the demand curve). Changes in consumer income, prices of related goods, consumer tastes, or most importantly for the present discussion, the quality of the good can cause shifts in demand. For example, an improvement in quality would increase demand, shifting it to the right (from D_1 to D_2 as shown in Figure 1). When the demand curve shifts to the right the associated consumer surplus area increases (DCS). This change in consumer surplus is the change in net economic benefits from the quality improvement. In practice, changes in consumer surplus have been found to be good approximations of more theoretically correct measures of economic benefit (Willig 1976; Randall and Stoll 1980). See Johansson (1987) for additional detail on the theory of environmental valuation.

Estimation of consumer surplus is relatively straightforward if market data exist. Typically, the demand curve equation is estimated statistically using data on market prices, quantities purchased by consumers, and other related variables such as consumer incomes and prices of related goods. Without market data, a number of methodologies have been developed to estimate consumer surplus. Consumer sur-
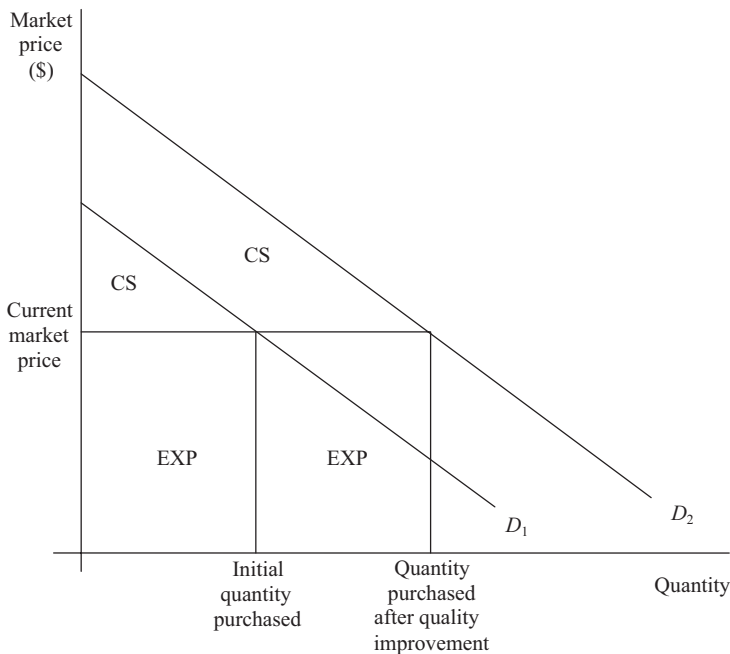
FIGURE 1. Relations among demand (*D*), consumer surplus (CS), and expenditure (EXP).

plus for nonmarket goods such as water quality improvements can arise from two sources: *use value* and *nonuse value*. Both use and nonuse values can be estimated using direct and indirect methodologies, although the latter are typically better suited for the estimation of use values, while the former are better suited for estimating nonuse values.

Indirect "Revealed Preference" Methods

The travel cost method (Bockstael 1995) is a revealed preference method that is most often used to estimate the benefits of outdoor recreation (e.g., improved fishing opportunities following water quality improvement). The travel cost method begins with the insight that the major cost of outdoor recreation is the travel and time costs incurred to get to the recreation site. Since individuals reside at varying distances from the recreation site, the variation in distance and the number of trips taken are used to trace out a demand curve for the recreation site. The demand curve is then used to derive the consumer surplus associated with using the site. With data on appropriate demand curve shift variables (i.e., independent variables such as measures of water quality), the economic benefits (i.e., changes in consumer surplus) associated with changes in the

shift variables (i.e., changes in water quality) can be derived.

A variation of the travel cost method is the random utility model (RUM) (e.g., Bockstael et al. 1989). Unlike the traditional travel cost model which focuses on one recreation site, a RUM uses information from multiple recreation sites. Individuals choose a recreation site based on differences in trip costs and site characteristics (e.g., water quality) between the alternative sites. Statistical analysis of the relationship between site characteristics and recreationists' site choices enables estimation of any consumer surplus changes arising from any changes in site characteristics, such as water quality.

The averting behavior method (Smith 1991) begins with the recognition that individuals seek to protect themselves when faced with environmental risk such as contaminated drinking water. Defensive behavior requires expenditures that would not normally be made. For example, purchases of bottled water or water filters may increase when the risk of contaminated drinking water increases. These increases in expenditures represent a lower bound on the economic benefits of policy that reduces drinking water risk.

The hedonic price method (Palmquist 1991; Freeman 1993) exploits the relationship between charac-

teristics of land and labor markets, including water quality, and housing prices and wages. For example, land parcels in close proximity to water bodies with high quality water command higher prices than parcels adjacent to water with lower quality. Job markets with greater environmental amenities (such as high quality water) are associated with lower wages relative to other job markets because individuals are willing to accept lower wages in order to gain greater amenities. Housing and labor market differences can therefore be used to trace out the demand for water quality and used to measure economic benefits.

The travel cost, averting behavior, and hedonic methods are considered indirect valuation methods because they estimate the benefits of water quality improvement (or other nonmarket goods) through an examination of demands for related goods such as recreational trips and housing. The major strength of indirect approaches is that they are based on data reflecting actual market choices, where individuals bear the actual costs and benefits of their actions. However, indirect methods are generally only suitable for the estimation of use value, as nonuse value may not be reflected in market choices and behavior. The major weakness of indirect approaches is their reliance on historical data. Policies often are beyond the range of historical experience. For example, few residents of an urban area located near a long-degraded stream may have experienced a fishable stream. Without variation in the historical water quality data, it is difficult to predict how an improvement in water quality would shift the residents' demand curve and change their consumer surplus. Analysis of the economic benefits of water quality policy is often difficult when indirect valuation methods are used exclusively.

Direct "Stated Preference" Methods

The contingent valuation method (Mitchell and Carson 1989; Bateman and Willis 1999) is a stated preference approach that directly elicits willingness (and ability) to pay statements from survey respondents. In other words, respondents are directly asked about their willingness to pay (i.e., change in consumer surplus) for environmental improvement or willingness to accept (i.e., amount of monetary compensation required to allow) environmental degradation.

The method involves the development of a hypothetical market via in-person, telephone, mail, or other types of surveys. In the hypothetical market respondents are informed about the current prob-lem and the policy designed to mitigate the problem. The state of the environment before and after the policy is described. Other contextual details about the policy are provided such as the policy implementation rule (e.g., majority voting) and the payment vehicle (e.g., increased taxes or utility bills). Finally, a hypothetical question is presented that asks respondents to choose between improved water quality with increased costs or the status quo. The choice is often framed as a referendum vote in order to make the situation more realistic. Respondents can be presented with multiple scenarios and make multiple choices. Statistical analysis of these data leads to the development of willingness to pay and consumer surplus estimates.

The contingent behavior approach is similar to the contingent valuation method in that it involves hypothetical questions. In contrast, the questions involve changes in hypothetical behavior instead of hypothetical changes in willingness to pay. For example, respondents can be asked about hypothetical recreation trips with and without water quality improvements (Whitehead et al. 2000). Conjoint analysis is a type of contingent behavior approach that asks about hypothetical recreation site choice and other discrete choices (Louviere 1988; Adamowicz et al. 1999). Again, respondents can be presented with multiple scenarios and make multiple choices. Contingent behavior and conjoint analysis responses are treated as behavioral data and are analyzed using the same statistical methods as are used in the indirect approaches.

A strength of the direct or stated preference approaches is their flexibility. Water quality policies are often new policies with no historical precedent. Absent a natural policy experiment, the historical (i.e., revealed preference) data does not contain observations related to the policy. Direct approaches can be used to construct realistic policy scenarios for any new policy. Oftentimes, hypothetical choices are the only way to gain policy relevant nonmarket benefit information. Another strength of the direct approaches, especially contingent valuation, is the ability to measure nonuse values, such as the value of improving aquatic ecosystems. The major weakness of the direct approaches is their hypothetical nature. Respondents are placed in unfamiliar situations in which complete information may not be available. At best, respondents give truthful answers that are limited only by their unfamiliarity. At worst, respondents give unconsidered answers due to the hypothetical nature of the scenario.

Benefit Transfer

The benefit transfer approach to environmental valuation was developed for situations in which the time and/or money costs of primary data collection for original direct and indirect studies are prohibitive. With benefit transfer, environmental benefit estimates from existing case studies (i.e., the study sites) are spatially and/or temporally transferred to a new, policy case study (i.e., the policy site). The more common type of benefit transfer is the spatial transfer, where consumer surplus from the study site is transferred to the policy site at the same point in time. Less common is the temporal transfer in which consumer surplus from one time period is transferred to another time period.

Benefit transfer has been widely used to inform policy analysis since the 1950s (Smith 1992; Bergstrom and DeCivita 1999). Yet, it was not until a 1992 special issue of *Water Resources Research* that attention was focused on the theory and practice of benefit transfer (Brookshire and Neil 1992). Research focusing on benefit transfer has rapidly increased since the special issue. Four benefit transfer methodologies have emerged: benefit estimate transfer, benefit function transfer, meta-analysis transfer, and most recently, preference calibration transfer. Each of these transfer methodologies can be used to transfer benefit estimates obtained from a variety of benefit estimation methodologies, such as travel cost, contingent valuation, and hedonic valuation.

Brouwer (2000) proposes some necessary conditions for a valid benefit transfer. First, consumer surplus from the study site must be theoretically and methodologically valid. Second, the populations in the study and policy sites must be similar. Third, the difference between prepolicy and postpolicy quality (or quantity) levels must be similar across study and policy sites. Fourth, the study and policy sites must be similar in terms of environmental characteristics. Fifth, the distribution of property rights and other institutions must be similar across sites. Accuracy of benefit transfer will suffer if any of these conditions is violated. Yet, as will be shown below, the degree to which accuracy is impacted depends greatly upon the measures used and the assumptions made.

Benefit Estimate and Function Transfer

Benefit function transfer should be distinguished from benefit estimate transfer. Benefit *estimate* transfer uses environmental benefit estimates developed for a study site at the policy site. Researchers simply obtain a benefit estimate from a similar study conducted elsewhere and use it for the current policy analysis case study (e.g., Luken et al. 1992). In contrast, benefit *function* transfer uses a statistical model of benefits developed at the study site to estimate benefits at the policy site (e.g., Desvousges et al. 1992). Characteristics from the policy site are substituted into the model from the study site to tailor benefit estimates for the policy site.

Loomis (1992) argues that benefit function transfer can be more powerful than benefit estimate transfer in situations where demographic or environmental quality factors (for example) at the study site differ from those at the policy site. However, empirical results concerning the superiority of benefit function transfer are mixed. In a study of Wisconsin lake recreation, Parsons and Kealy (1994) find that benefit function transfer estimates are within 4% of the original model estimates, while benefit estimate transfers are within 34%. Brouwer and Spaninks (1999) also find that benefit function transfer is more accurate (within 22%) than benefit estimate transfer. Loomis (1992) finds that recreational fishing benefits developed using the travel cost method transfer from one state to another with between 5% and 15% accuracy. Loomis et al. (1995) find that per capita reservoir recreation benefit estimates from a travel cost model transfer accurately across sites.

In contrast, Barton (2002) finds that benefit estimate transfer, with transfer errors of 20% and 30%, outperforms benefit function transfer in the case of water quality improvements in Costa Rica. In a study of marine recreational fishing using the contingent valuation method, Downing and Ozuna (1996) find that few benefit functions transfer and, of those that do, few benefit estimates generated from the benefit functions transfer accurately. Similarly, in a study of recreation sites in Arizona and New Mexico using contingent valuation, Kirchhoff et al. (1997) find that between 55% and 90% of the benefit function transfer estimates are not accurate.

Meta-Analysis Transfer

Meta-analysis is a general term for any methodology that summarizes results from several studies. In the case of environmental benefit transfer, benefit estimates gathered from several studies serve as the dependent variable in regression analysis, and characteristics of the individual studies (e.g., water quality, type of survey methodology) serve as the independent vari-

ables. Benefit transfer using meta-analysis has three advantages over benefit function transfer (Shrestha and Loomis 2001). First, by employing a large number of studies, benefit estimates will be more rigorous. Second, meta-analysis may be used to control for differences in functional form and other methodological differences across studies (Smith and Kaoru 1990a). Third, differences between the study site and the policy site can be better controlled.

Several meta-analysis studies focus on one valuation method and one type of environmental commodity. Smith and Kaoru (1990a, 1990b) conducted a meta-analysis of the benefit estimates derived from travel cost recreation demand models. Smith and Huang (1993, 1995) conducted a meta-analysis of air quality benefits derived from hedonic property value models. These studies confirm that study methodology influences benefit transfer estimates. The authors recommend that meta-analysis be used as a complement to other benefits transfer methods. Smith and Osborne (1996) conducted a meta-analysis of air visibility benefits. They found that benefit estimates tend to conform to important economic principles that confirm their validity, but this conclusion is subject to variation in research methods used in the studies. Loomis and White (1996) conducted a meta-analysis of studies of rare and endangered species. Their model is able to explain more than 50% of the variation in these values. They conclude that meta-analysis is a promising technique for benefit transfer.

Two meta-analysis studies compare alternative environmental valuation methods for a single environmental commodity. Walsh et al. (1992) conducted a meta-analysis of outdoor recreation value estimates from travel cost and contingent valuation studies. Woodward and Wui (2001) conducted a meta-analysis of studies of wetland values using travel cost, contingent valuation and other methods. Both studies conclude that the contingent valuation method tends to generate lower benefit estimates relative to other methods. A similar result is found by Carson et al. (1996).

Rosenberger and Loomis (2000) compare national and census region meta-analysis functions. The national and census region models produce benefit estimates that differ from those in the original studies by 54% and 71%, respectively. Benefit transfers are more accurate for activities with many existing studies in the database, such as fishing, than for activities with only a few studies, such as skiing. Shrestha and Loomis (2001) use results from U.S. studies to forecast benefits for international policy sites. They find

that average prediction error is between 24% and 30% after adjusting for inflation and exchange rates.

Finally, Smith and Pattanayak (2002) provide a review of the meta-analysis literature. They argue that few existing meta-analyses should be used for benefit transfer due to inconsistent definitions of the benefit estimates (e.g., pooling estimates from contingent valuation and travel cost methods) and environmental commodities (e.g., value derived for use versus nonuse values).

Preference Calibration Transfer

Smith et al. (2002) and Pattanayak et al. (in press) argue that a new approach to benefit transfer, preference calibration, is needed because the majority of the evidence appears to indicate that benefit function transfer is not accurate. As with benefit function transfer, preference calibration exploits benefit estimates from other studies. In contrast, preference calibration uses estimates from multiple methods to develop a preference function consistent with economic theory. Importantly, preference calibration ensures that benefit estimates do not violate the consumer's ability to pay requirement when the scale of the environmental change is large. In other words, preference calibration ensures that consumers can afford to pay the amounts indicated by the transferred willingness to pay estimates.

Smith et al. (2002) used preference calibration to estimate the benefits of improved water quality using contingent valuation, travel cost demand, and hedonic property value studies. They found that conventional benefit estimate transfer understate benefits by 83% for the travel cost studies and 3% for the hedonic property value studies. Conventional transfer overstate benefits by 64% for the contingent valuation study. Pattanayak et al. (in press) found that conventional benefit estimate transfer understate water quality benefits by 66% for travel cost studies and 16% for contingent valuation studies. The contingent valuation method performs better in the second study because it includes nonuse values as well as use values.

An Assessment

Three preferred types of benefit transfer are emerging: benefit function transfer, meta-analysis transfer, and preference calibration. Meta-analysis transfer has several advantages over benefit function transfer. A major advantage is that meta-analysis is able to control for differences in study methodologies. However, meta-

analysis suffers from (1) reporting errors and omissions in the original studies, (2) inconsistent definitions of environmental commodities and values, and (3) large random errors. In addition, the development of a meta-analysis function is costly in terms of time and money relative to benefit function transfer due to the larger number of studies required.

Preference calibration has been proposed as a solution to the problems associated with benefit function transfer and meta-analysis transfer. A major benefit of preference calibration is its recognition that willingness to pay is constrained by income in situations involving large changes in policy variables. However, there are several problems with preference calibration. Preference calibration does not tailor the benefit estimates to the demographics and other characteristics of the policy site as does benefit function transfer and meta-analysis transfer. Preference calibration is more time consuming than benefits function transfer due to the increased analytical burden. Also, preference calibration has yet to be vetted by tests of transfer accuracy.

Numerous and restrictive conditions are necessary for the successful application of each of the three emerging benefit transfer methods. It is not surprising that many studies evaluating benefit transfer methods reject transfer accuracy. In other words, the differences between benefits from a primary study and transferred benefits are statistically significant. Nonetheless, the benefits from a primary study and transferred benefits are typically of the same order of magnitude and differences are typically much less than 100%. When primary data collection is not feasible, there are no current alternatives to benefit transfer. The practice of benefit transfer is sure to continue.

Policy Study: Cape Fear River

In this section, we use benefit transfer methods to estimate the benefits of hypothetical water quality improvement policies for residents of an urban area. Although it would be an interesting methodological exercise to estimate the benefits of a water quality improvement *for the same policy* using alternative benefit transfer methods to test their validity, the purpose of this paper is to illustrate the empirical use of existing methods. Given the limited scope of this study, we do not employ the time-intensive meta-analysis or preference calibration approaches to benefit transfer. Instead, we apply the benefit estimate and benefit function transfer approaches using the travel cost, hedonic price, and contingent valuation methods of estimating willingness to pay and consumer surplus. The analysis illustrates how the benefit transfer approaches are used in combination with the valuation methods to obtain benefit estimates.

The case study site is the portion of the lower Cape Fear River that flows through New Hanover County, located in the southeastern corner of North Carolina (Figure 2). The Cape Fear River basin is the largest river basin in North Carolina (North Carolina Department of Environment and Natural Resources 2000). It originates near Greensboro and flows east past the Chapel Hill-Durham area and southeast to Wilmington (population = 75,838) in New Hanover County (population = 165,712) where it drains into the Atlantic Ocean. The Cape Fear River basin is comprised of the Haw, Deep, upper Cape Fear, Black, northeast Cape Fear, and lower Cape Fear watersheds.

The Cape Fear River is subject to point-source water pollution from industrial and municipal waste treatment facilities and nonpoint source pollution from agricultural runoff, storm water runoff from urban and suburban areas, and sediment from newly urbanizing areas. As of 1999, there were 280 point-sources of wastewater in the Cape Fear River basin permitted under the National Pollutant Discharge Elimination System (NPDES), with a total permitted flow of 1.34 million m³/d (353 million gallons/d, MGD) (North Carolina Department of Environment and Natural Resources 2000). Of these, 58 were major sources, each emitting more than 3,700 m³/d (1 MGD). The lower Cape Fear contains more than 50% of the agricultural hog production operations in North Carolina. Nutrients from treated hog waste sprayed onto field crops as fertilizer flow into tributary waters during high rainfall events. Although one-half of the land area is forested, the Cape Fear River basin is a rapidly urbanizing area. For example, Wilmington experienced significant economic growth during the 1990s, its population increasing by 29.4%. Land clearing and construction activities associated with development increase the sediment load in the river. As of 1999, 623 general stormwater permits (typically construction projects affecting two or more hectares) and forty-eight individual (large municipal and industrial) stormwater permits were issued within the basin under the stormwater program of the 1990 Clean Water Act.

Multiparameter water quality sampling for the Cape Fear River has been conducted by the Lower Cape Fear River Program (LCFRP) since June 1995 (Mallin et al. 2002). The LCFRP currently encompasses 35 water sampling stations throughout the Cape
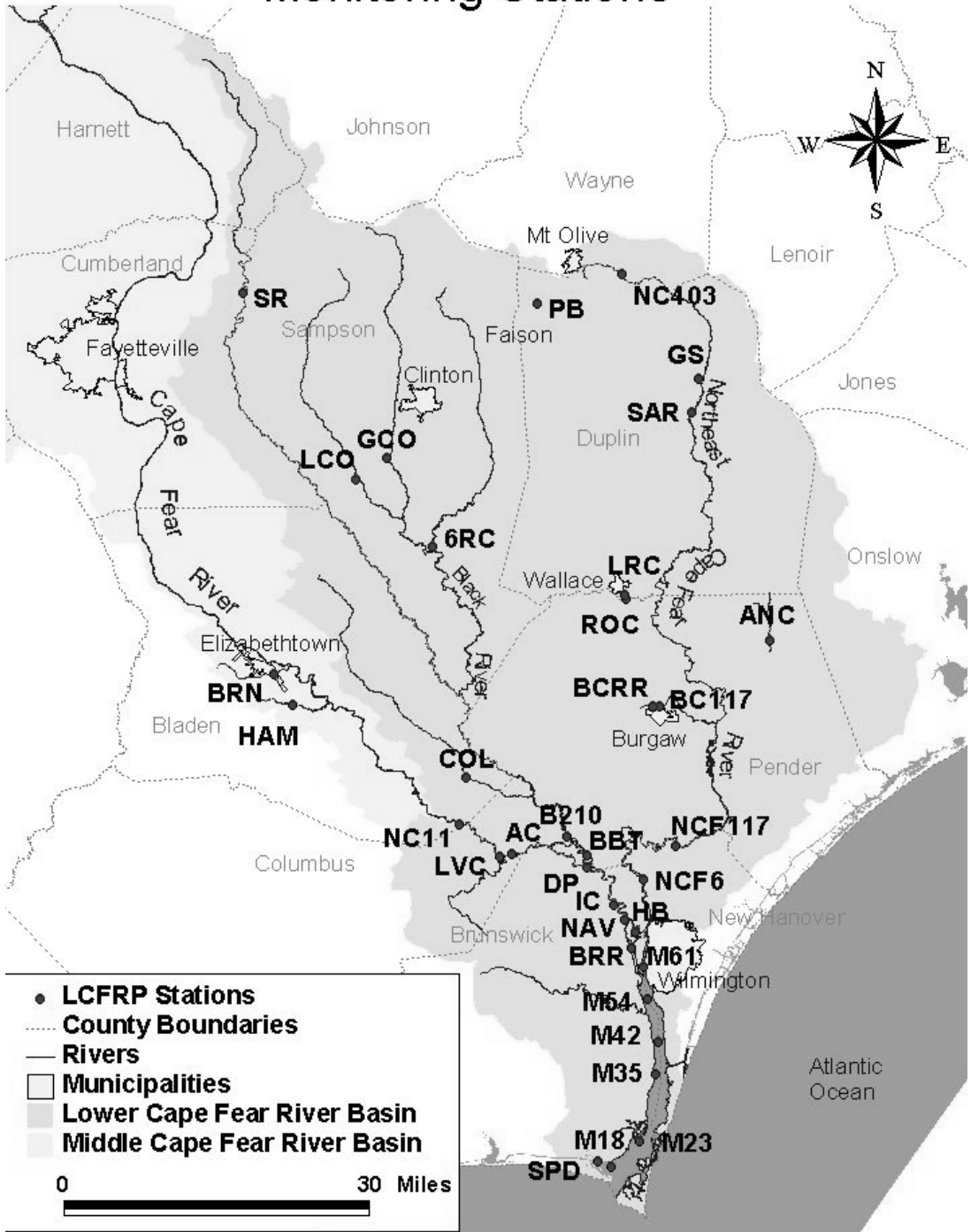
FIGURE 2. The Cape Fear River basin, North Carolina, showing Lower Cape Fear River Program water quality sampling station locations (Lower Cape Fear River Program 2003).

Fear, Black, and northeast Cape Fear River watersheds. The LCFRP sampling program includes physical, chemical, and biological water quality measure-ments, analyses of the benthic and epibenthic macroinvertebrate assemblages, and assessment of the fish assemblages.

The main-stem lower Cape Fear River is characterized by somewhat turbid water containing high levels of inorganic nutrients. It is fed by two large blackwater rivers (the Black and northeast Cape Fear rivers) that have low levels of turbidity, but darkly colored water (due to naturally occurring tannins), with less inorganic nutrient content than the main stem. While nutrients are reasonably high in the river channels, algal blooms are rare because light is attenuated by water color or turbidity and flushing is high. Periodic algal blooms are seen in the tributary stream stations, some of which are impacted by point source discharges. Below some point sources, nutrient loading can be high and fecal coliform contamination occurs. Other stream stations drain blackwater swamps or agricultural areas, some of which periodically show elevated pollutant loads or effects.

During the 2001–2002 sampling period, a prolonged drought had a significant positive effect upon water quality. As a result of the drought conditions, a considerably lower number of stations were impaired by fecal coliform contamination than in the past several years. The impaired locations were a mixture of areas impacted by point and nonpoint source inputs. Against this background, we estimate the benefits of water quality improvement with the benefit transfer approach.

Benefit Estimate Transfer: Travel Cost Method

To illustrate a temporal benefit estimate transfer using the travel cost method of valuation, we apply estimates of the benefits of ambient water quality improvements in river basins and watersheds in North Carolina from Phaneuf (2002). Phaneuf (2002) used data from the Environmental Protection Agency's (EPA) national water-based recreational survey, which are combined with chemical measures of water quality. The random utility model (RUM) version of the travel cost method is employed to model behavioral responses to changes in water quality in order to aid in the design and implementation of total maximum daily load (TMDL) policies in North Carolina. As noted above, given that travel costs serve as an implicit price of a recreation visit, changes in

recreational site choices in response to changes in water quality can be used to estimate the use value of water quality improvements.

Phaneuf (2002) estimated the benefits of four potential changes: the loss of individual watersheds from recreation use, water quality improvements in individual watersheds, water quality improvements across an entire river basin, and reductions in ammonia and phosphorous. The specific water quality improvement for the second of these measures is defined as a reduction in pollution loadings such that a maximum of 10% of monitoring station readings for pH, dissolved oxygen, phosphorous, and ammonia are out of compliance for the watershed and is most applicable for our purposes here—to illustrate benefits transfer for a specific watershed. In addition to quantifying the value of reductions in pollutant loadings using individual measures of the pollutants, Phaneuf (2002) also derived the willingness to pay for the same improvements as measured by the EPA's index of watershed indicators (IWI) (U.S. Environmental Protection Agency 2002). This index is a scale of 1–6, with 1 indicating the highest water quality.

For the watersheds in the Cape Fear River basin, willingness to pay per trip to maintain (i.e., to prevent the loss of) existing recreation access is $0.29 for the upper Cape Fear River, $0.39 for the lower Cape Fear River, and $0.80 for the northeast Cape Fear River (Phaneuf 2002). Further, the willingness to pay per trip for the water quality *improvement* was found to be $0.10 for the upper and lower Cape Fear River and $0.24 for the northeast Cape Fear River. The mean willingness to pay per trip estimates across all watersheds in the state were $0.41 for access and $0.17 for the improvement. The ranges of these estimates were $0.05 to $2.91 and $0.00 to $1.44.

Phaneuf (2002) found that the per trip willingness to pay for a reduction in pollution loadings such that a maximum of 10% of readings are out of criteria for the entire Cape Fear River basin (as opposed to a single watershed within the basin) are between $1.00 and $6.29, depending on the specification of the statistical model and which water quality data are used. The per trip willingness to pay value found using the IWI is $2.25 (Phaneuf 2002). In terms of the benefits transfer, a lower bound on the aggregate benefits of basin-wide improvements over the entire season is approximated by multiplying these per trip benefits by the total number of freshwater angling days in North Carolina. The U.S. Fish and Wildlife Service estimated that 675,000 resident anglers fished 11.4 million freshwater days in North Carolina in 2001

(U.S. Department of the Interior, Fish and Wildlife Service and U.S. Department of Commerce, U.S. Census Bureau 2002). These estimates lead to an aggregate value of $31.8 million (2003 dollars) for the basin-wide water quality improvements using the IWI estimate. Using the range of values estimated for the 10% out-of-criteria improvements, this annual aggregate benefit measure is between $14.1 million and $88.9 million (2003 dollars).

We also obtain an aggregate estimate for New Hanover County by using data on North Carolina freshwater angler-days and population estimates for the state and county (New Hanover County contains approximately 2% of the North Carolina population). Assuming that the proportion of anglers in the population is constant across counties, this amounts to 13,500 resident anglers fishing 228,000 freshwater days in New Hanover County. These estimates lead to an aggregate value of $636,000 (2003 dollars) for the basin-wide water quality improvements using the IWI estimate. Using the range of values estimated for the 10% out-of-criteria improvements, the value to New Hanover County anglers is between approximately $283,000 and $1.86 million (2003 dollars).

Benefit Function Transfer: Hedonic Price Method

The existing hedonic studies of the value of water quality typically use water clarity or fecal coliform as a measure of water quality. We select fecal coliform, a group of bacteria widely used as an indicator of the presence of disease-producing bacteria, as our measure of water quality for the hedonic analysis. Water clarity would not be a good measure of water quality for the New Hanover county area, as several tributaries of the Cape Fear River are naturally low-visibility, low-clarity waters in their pristine states (due to naturally occurring tannins in the water). Fecal coliform measurements vary by an order of magnitude above and below the state health standard for human contact waters (200 CFU/100 mL) in the Lower Cape Fear River. During the 2001–2002 monitoring period, the state standard was exceeded six times (North Carolina Department of Environment, Health, and Natural Resources 1996). (The standard is typically violated more frequently; the 2001–2002 period had a relatively low number of violations due to low run-off conditions during a drought.)

For the benefit transfer application, we select Leggett and Bockstael's (2000) hedonic pricing study of the effect of fecal coliform water pollution on Chesa-

peake Bay shoreside property values. In addition to its focus on fecal coliform pollution, Leggett and Bockstael (2000) utilized relatively recent data (late 1990s) and considered coastal estuarine properties in the mid-Atlantic region of the United States, properties similar to those in our study region. We recalibrate the Leggett and Bockstael (2000) hedonic price model to New Hanover conditions. The recalibration accounts for differences in parcel area, distance to urban centers, and baseline fecal coliform levels between the Leggett and Bockstael (2000) study area and New Hanover County. The model is not recalibrated for differences between the two study areas in neighborhood land uses or distances to point sources of water pollution. For these variables, we use the mean values from Leggett and Bockstael (2000).

Land parcel and tax data for 2001 were provided by the New Hanover County Planning Department. Industrial, government, commercial and utility right-of-way parcels are excluded from the analysis. The remaining 334 residential and residential/farm parcels adjacent to the Cape Fear and northeast Cape Fear rivers in New Hanover County in 2001 occupy a total of 3,554 ha. The mean land value per parcel (excluding the value of any structures) is approximately $121,000 for residential land use ($n = 331$) and $300,000 for residential/farm land use ($n = 3$). Fecal coliform is measured at LCFRP water quality monitoring field station NAV, just north (upstream) of Wilmington, North Carolina (see Figure 2). From 1997–2002, monthly average fecal coliform readings varied from a minimum of 6 CFU/100 mL to a maximum of 4,453 CFU/100 mL, depending on season, rainfall, and point source and nonpoint source pollution discharges, with a geometric mean of 31 CFU/100 mL.

The policy scenario consists of a hypothetical water quality program that would prevent deterioration of water quality from a baseline yearly median fecal coliform count of 40 CFU/100 mL, a level approximating current conditions, to the level of the state health standard for human contact waters, 200 CFU/100 mL. Using the Leggett and Bockstael (2000) model recalibrated for the Cape Fear region, we found that the 334 riverfront residential properties in New Hanover County have an aggregate land value (excluding the value of any structures) of approximately $42.4 million (2003 dollars) under baseline water quality conditions of 40 CFU/100 mL. If water quality were allowed to deteriorate to the level of the state health standard for human contact waters (200 CFU/100 mL), land value would fall to an esti-

mated level of $39.1 million, a loss of $3.3 million. This is equivalent to a 7.7% decrease in land value. The maximum decrease in value for any single property is $510,000 (for a 538-ha parcel slated for subdivision), the minimum decrease is $12, the mean decrease is $9,800, and the median decrease is $4,400.

Benefit Function Transfer: Contingent Valuation Method

The contingent valuation method literature contains a number of studies that estimate the economic values of river water quality. Several of these are focused on North Carolina river basins, but none focuses on the Cape Fear River basin. A recent study estimated the economic value of water quality protection in the Catawba River basin (Kramer and Eisen-Hecht 2002). The Catawba River basin is similar to the Cape Fear River basin in that it originates near an urban area, Charlotte, and flows southeast to the Atlantic coast. It differs in that the Catawba River basin is dominated by reservoirs and most of the basin is located in South Carolina. Nevertheless, we choose this as the study site due to its similarities to the policy site and the richness of the statistical valuation function relative to other North Carolina river basin valuation studies.

Kramer and Eisen-Hecht (2002) used a combination of mail and telephone survey methods. The sample is mailed an information booklet that describes a water quality management plan for the Catawba River. The booklet includes maps that show the potential deterioration in water quality given current population and land use changes as predicted by a water quality model. The proposed management plan would focus on several water quality problems: sediment, nutrients, toxic substances, bacteria, and viruses. The management plan would include the use of best management practices for construction and agriculture within the basin, develop a basin-wide land use plan, improve and increase the capacity of sewage treatment plants within the basin, and provide for the purchase and protection of land that is important for the protection of water quality.

Respondents were asked to vote for or against the management plan given that it would be financed by a specified increase in state income taxes over the following 5 years. The specified increase in state income taxes varied across survey respondents, ranging from $5 to $250 per year. Without further water quality information, the contingent valuation method cannot be used to place a monetary value on a specific water quality improvement (i.e., a change in pH or

fecal coliform units). The benefit estimate from this application of the contingent valuation method is the willingness to pay for protection of current water quality with the proposed water quality management plan. Additional information from the water quality model that was used to estimate the potential degradation in water quality could be used to develop estimates for specific improvements. However, this level of analysis is beyond the scope of this paper.

Kramer and Eisen-Hecht (2002) statistically analyzed the survey data to develop a willingness to pay model. The model includes a number of variables that can be used to examine the validity of the hypothetical votes. For example, the probability of a vote for the management plan should fall as the tax amount increases and should rise with increases in respondent income. Such results were obtained in this study and indicate that respondents responded rationally to the stated cost of the policy relative to their income levels. These results strongly suggest that the hypothetical votes reveal valid economic values for Catawba River water quality.

Kramer and Eisen-Hecht (2002) estimated that respondent annual willingness to pay is $194 (1998 $) for 5 years for the Catawba River. The Catawba River willingness to pay model is calibrated for New Hanover County residents. Calibration involves substitution of relevant values from the policy site (New Hanover County) for the values used in the study site (Catawba River basin).

There are no objective measures for New Hanover County residents for most variables in the willingness to pay model. For these variables, we used the mean values from the Catawba River basin sample (Eisen-Hecht and Kramer 2002). These include study specific variables, such as knowledge and attitudes about water quality, and variables specific to the survey design. The willingness to pay also includes a variable for whether the respondent is from North Carolina or South Carolina. We set this variable equal to South Carolina, assuming that downstream New Hanover County residents are more similar to respondents in South Carolina than the upstream, urban North Carolina respondents. This choice has significant effects on willingness to pay. The alternative assumption would decrease annual willingness to pay estimates by almost $62.

For the demographic variables measuring respondent age, education, sex, and household income, we developed estimates of the mean values for New Hanover County residents 18 years or older using U.S. Census Bureau data. We assume that respondents would rate the use of the river as important and

that drinking water is important. In order to differentiate between use and nonuse values, we alternatively assumed that altruistic, bequest, and existence values are zero and positive. The means from the Catawba River sample are used for all other variables.

Assuming that the willingness to pay functions for the Catawba River and the Cape Fear River are similar, these estimates represent the willingness to pay of New Hanover County residents to maintain water quality through a Cape Fear River basin-wide management plan. The willingness to pay of New Hanover County households is $175 per person, per year, for 5 years when nonuse values are equal to zero and $326 per person, per year, for 5 years when nonuse values are positive (2003 dollars). When nonuse value is considered the residual between total value and use value, this implies that nonuse values are 46% of the total value.

Comparison of Methods

The benefits of water quality improvement in the Lower Cape Fear River varied with method (Table 1). Willingness to pay estimates developed from the travel cost, hedonic price, and contingent valuation methods are aggregated by the number of New Hanover County angler-days (n = 13,500), New Hanover County residential properties in vicinity of the Cape Fear River (n = 334 properties), and New Hanover County households (n = 68,183), respectively. The raw value estimates from the transfer studies are not directly comparable for two reasons. First, the estimates refer to different time periods: the contingent

valuation estimates are annual values for each of 5 years, the travel cost estimates are annual values received each year in perpetuity, and the hedonic price method estimate is a capitalized, present value. To make the estimates comparable, we calculated the present value of the annual amounts (using a 5-year time horizon for the contingent valuation estimates and a 30 time horizon for the travel cost method), and we annualized the hedonic price method estimate.

Second, each benefit transfer example focuses on a different policy context. The travel cost method willingness to pay estimate is appropriate for a policy that leads to a reduction in pollution loadings such that a maximum of 10% of readings are out of criteria for the entire Cape Fear River basin (as measured by a one unit change in a water quality index). In contrast, the hedonic price method and contingent valuation method estimates are appropriate for a water quality management plan that protects the current level of water quality, though the two estimates are based on different definitions of the current level of water quality and different definitions of the water quality management plan.

We used two discount rates for the present value calculations. The first discount rate, 2%, is a frequently used approximation of the real discount rate based on market interest rates and is recommended by the Congressional Budget Office (Hartman 1990). The second and higher discount rate, 7%, is required for benefit–cost analysis by the U.S. Office of Management and Budget (Office of Management and Budget 1992). The higher rate is based on the

TABLE 1. Aggregate benefits for Lower Cape Fear River water quality (millions of 2003 dollars).

Method	Aggregation	Policy	Annual			Present value		
			Unad-justed	CBO[1]	OMB[2]	Unad-justed	CBO[1]	OMB[2]
Travel cost	228,000 angler-days	Avoidance of 10% of water quality monitoring stations being out of compliance	$0.64				$14.33	$7.94
Hedonic price	334 proper-ties	Protection of water quality to avoid increase in fecal coliform from current level		$0.15	$0.27	$3.30		
Contin-gent valua-tion	68,183 house-holds	Protection of current water quality with a water quality management plan	$22.20				$104.62	$91.01

[1] Value adjusted based on the Congressional Budget Office discount rate of 2%.
[2] Value adjusted based on the U.S. Office of Management and Budget discount rate of 7%.

market rate of return of housing and corporate borrowing costs.

With discount rates of 2% and 7% the present value of aggregate benefits for anglers using the travel cost method are $14 million and $8 million. The hedonic price method gives the present value (capitalized value) of aggregate benefits for *riverfront* property owners directly; this value is $3.30 million. Using the hedonic price method estimate of $3.30 million and discount rates of 2% and 7%, the annualized value of the aggregate benefits for property owners are $0.15 million per year and $0.27 million per year. Using the contingent valuation method and discount rates of 2% and 7%, the present value of aggregate benefits (total value including nonuse value) for all households in the county (not just riverfront) are $105 million and $91 million.

This comparison illustrates the limitations of the alternative methods. The travel cost and hedonic price methods are applicable to particular populations and are not able to measure nonuse values. The contingent valuation method can be used to estimate nonuse values and is applicable to the entire population that might enjoy nonuse values. However, it is difficult to disentangle use and nonuse values from the total value estimate with the contingent valuation method.

It is tempting to add the estimates from the three methods to generate an estimate of the total benefit of the water quality improvement. However, this temptation is misguided for two reasons. First, the benefit estimates are for different policies as described above. Second, the total benefit estimate would be prone to double counting of benefits. The travel cost method primarily estimates the water quality benefits that are enjoyed by those who participate in outdoor recreation. The hedonic price method estimates the benefits of water quality improvements that accrue to property owners. Since proximity to recreation sites is an incentive for property owners to purchase housing near water, the benefits accruing to property owners might include recreation benefits. The contingent valuation method estimates the use values, including recreation benefits, for the general population. Adding the benefits from the travel cost method, the hedonic price method, and the contingent valuation method might include recreation benefits for three overlapping populations.

Summary

In this paper, we provide an accessible primer on the economics of water quality valuation. Consumer surplus, the net benefits of a particular good, can be estimated using a number of valuation methodologies, including direct and indirect methods. These methods typically require the collection of new data. Yet, policy analysis is often constrained by time and money. In these situations, benefit transfer methods can be used to develop estimates of consumer surplus for policy analysis. Benefit transfer involves the recalibration of existing consumer surplus estimates. Existing estimates are tailored to fit a new policy situation. We provide an example of benefit transfer by estimating the value of water quality improvements for the Cape Fear River in North Carolina. Benefit transfer methods are used with three valuation approaches (travel cost, hedonic pricing, and contingent valuation) to estimate the benefits of water quality improvements.

The successful application of benefit transfer methods remains a challenge. Brouwer (2000) provides some restrictive conditions for a successful benefit transfer. Many studies evaluating benefit transfer methods that adhere to most of Brouwer's (2000) conditions reject the statistical accuracy of benefit transfer estimates. However, benefit transfer methods typically obtain accuracy within an order of magnitude. The role of the benefit estimate in the policy process and the costs of a wrong decision are the two major issues that must be addressed when deciding whether to use a benefit transfer method instead of collecting primary data (Bergstrom and DeCivita 1999). Typically, benefit cost analysis is only one input into the policy decision process. When government water quality policy decisions do not hinge on whether the present value of net benefits is positive or negative, in other words, when the benefit cost analysis is advisory, the use of benefit transfer is a an acceptable approach to obtain order of magnitude estimates of benefits.

When major government decisions are made, such as reauthorization of the Clean Water Act, the costs of a wrong decision could be in the millions, or even billions, of dollars. When determining whether to conduct a study based on primary data, the cost of the study must be compared to the potential cost of a wrong decision. For example, a benefit cost analysis that uses benefit transfer to estimate benefits may conclude that the present value of net benefits of a policy is $2 million. Based on the criterion of efficiency, the policy analyst would recommend that the policy should be pursued. However, a benefit cost analysis that uses new, primary data to estimate benefits may conclude that the present value of net ben-

efits of the same policy is –$2 million. In this case, the policy analyst would recommend that the policy should not be pursued. If the study based on new, primary data costs $500,000, then it is an investment with a net gain of $1.5 million (i.e., the $0.5 million study prevents a $2 million mistake). In this case, the study based on new, primary data are preferred to benefit transfer. For most water quality policies the costs of a wrong decision are much smaller. In many of these cases, the benefit transfer approach may be preferred.

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