AN ASSESSMENT OF META-ANALYSIS BENEFIT TRANSFER METHODS FOR VALUING WETLAND ECOSYSTEM SERVICES IN US NATIONAL WILDLIFE REFUGES

by

DOUGLAS ARTHUR PATTON

(Under the Direction of John C. Bergstrom)

ABSTRACT

In this dissertation, I conduct an assessment of meta-analysis benefit transfer methods by examining both existing meta-analysis models and by constructing a novel meta-analysis database and estimator. The analysis of both a newly developed and published meta-analysis studies focuses on ecosystem services provided by wetlands in U.S. National Wildlife Refuges (NWRs). An application of the meta-analysis models to four case study NWRs provides empirical examples of how meta-analysis benefit transfer can be used to value water quality enhancements and flood control/storm protection ecosystem services in specific wetland landscapes.

In Chapter 3, A Monte Carlo simulation and forecast combination of the dependent variable from three published meta-analysis models indicates that the models can be useful for rank ordering sites for low-stakes decision making or for preliminary assessments of ecosystem service values. However, the simulation also indicates that an assessment of the accuracy of the existing models cannot be determined without strong assumptions about the structure of each model’s error term covariance matrix.

A novel meta-analysis database in Chapter 4 with a similar focus to those assessed in Chapter 3 is developed in order to better quantify benefit transfer accuracy. Regression analysis of the meta-analysis database indicates similar rank orderings of predicted values relative to the results in the previous chapter. Advancements include more flexible modeling of local substitute wetlands and local populations as well as the inclusion of a user population variable omitted from earlier studies.

Chapter 5 develops a novel Parametric Locally Weighted Least Squares (PLWLS) estimator that empirically models the theoretical notion of *correspondence* that is often mentioned but has not yet been quantified in the ecosystem service valuation literature. The non-linear PLWLS estimator improves on previous meta-analysis models by establishing a systematic approach to resampling while improving the efficiency of benefit transfers. Employing a jackknife simulation to approximate resampling, I verify that the PLWLS method generates forecasts that are more accurate than the conventional modeling approach. The results also indicate that the PLWLS estimator in conjunction with the novel data set generates forecasts that are more precise than the forecasts from the three earlier meta-analysis models.

INDEX WORDS: Meta-Analysis, Meta-Analysis Benefit Transfer, Benefit Transfer, Non-Market Valuation, Wetland Ecosystem Services, Locally Weighted Regression

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CHAPTER 1

INTRODUCTION

Modern societies have transformed nature and developed technologies to create lifestyles that can seem quite distant from the influences of natural systems. However, regardless of one's perceived distance from Nature, there are a number of ways in which ecosystems support and enrich one's life. Economists theorize that people benefit from goods and services associated with various ecosystems in a manner that is comparable to the benefits obtained through conventional, market-priced goods and services. This theoretical framework allows economists to consider the trade-offs between goods and services obtained through the market economy and goods and services obtained by interacting with ecosystems. However, because markets do not provide information about prices for many of these ecosystem services, understanding the trade-offs associated with various landscape management strategies is an important topic for research (Daily 1997).

Due to the lack of market prices for many ecosystem services, signals are not well transmitted between those who demand ecosystem services and those who manage the ecosystems that supply these services. Consequently, quantification of the benefits to humans from ecosystem services allows for improved understanding and collective action that can enhance the quality and quantity of those beneficial ecosystem goods and services. This study seeks to improve scientific understanding of ecosystem service benefits through the development of low-cost valuation methods with the aim of improving decision making that affects the welfare of present and future generations. I specifically explore meta-analysis models of ecosystem service valuation studies. These meta-analysis models use quantitative techniques to identify patterns in a sample of studies that use primary data to assess non-market economic values.

The tools developed by economists to quantify the trade-offs between the widely enjoyed public goods provided by wetlands and the private goods (e.g., agricultural commodities) more readily produced after wetland conversion are useful for understanding how the public will be affected by changes in wetland ecosystem quantity and quality. These tools measure trade-offs in terms of dollar values, a process known as ecosystem service valuation. Importantly, the costs associated with using these non-market valuation tools have been a barrier to their widespread use by stakeholders and policymakers, especially for low-stakes decisions. Accordingly, there has been considerable interest in developing informative lower cost methods for understanding how people benefit from the ecosystem service flows supported by wetlands, in particular those service flows that are not traded in conventional markets.

Important questions remain unanswered regarding issues pertaining to the accuracy of information as well as the cost of information relating to wetland ecosystem service values. Low-cost yet accurate methods hold great promise for evaluating the individual and collective impact of the myriad land use decisions made by a variety of stakeholders with varying incentives. Due to the costs of conducting primary valuation studies, even substantial decisions regarding the use of public resources for ecosystem service conservation are typically made without formal benefit cost analysis that includes ecosystem service values. For decisions both large and small, systematically ignoring the important values of ecosystem services may lead to a decline in public welfare due to underprovision of these services.

The need for reliable value estimates that come at modest costs may be greatest for the large land conservation programs of the US federal government. For example, these values may be useful for guiding decisions about where to add new wildlife refuges and national forests or to evaluate if funding can be better used to expand existing ones. In this dissertation, I explore existing options that promise low-cost value estimates, but which have been questioned in terms of accuracy. Specifically, I analyze the performance of three meta-analysis studies based on summary information provided by the authors in peer-reviewed publications of their models as well as supplementary information occasionally available to guide the use of these models in valuation of wetlands and their ecosystem services. This use of existing valuation studies to predict economic values at unstudied sites is referred to in the literature as benefit transfer.

Because published meta-analysis model summaries contain insufficient information to fully understand the values the model can predict, I develop a meta-analysis database with a similar focus to several existing meta-analysis models, allowing for more in-depth analysis. I develop a novel statistical modeling approach that uses this database to efficiently forecast estimates of the benefits a population receives from different types and locations of wetland ecosystem,. This modeling approach formalizes a theoretical notion called *correspondence* while enhancing forecast precision without requiring the intervention of an analyst guided by experience and intuition. By avoiding this intervention, the new parametric locally weighted least squares (PLWLS) method reduces variability introduced by individual researchers. This approach also retains more information from an original sample that for benefit transfer targeted meta-analysis modeling would typically be narrowed down to a variable and small number of primary valuation studies.

Substantial challenges still remain for valid applications of meta-analysis benefit transfers. Diverse and precise ecological and socio-economic data are available for describing ecosystems and local populations but the data for describing the interactions between people and ecosystems are often lacking. Primary valuation studies are useful for describing these human-ecosystem interactions. However, the available primary valuation studies provide a relatively small number of observations for statistical analysis, covering a wide range of time periods, ecosystems, services, populations, and valuation methods. To date, meta-analysis models cannot fully account for all of the important features that distinguish populations that benefit from ecosystem services and the ecosystems that provide these services. Accordingly, these benefit transfer models are valid for answering questions about broadly defined ecosystem services. Because of the difficulties that exist in obtaining information about the economic nature of human-ecosystem interactions, novel primary valuation studies remain the best avenue for developing detailed information about the economic nature of human-ecosystem interactions.

This dissertation begins with a discussion of topics in the scientific literature that are relevant to wetlands and non-market valuation. Chapter 3 examines existing meta-analysis models with a Monte Carlo simulation and a forecast combination approach. An assessment of the barriers associated with using existing meta-analysis models is an important topic under-represented in the literature. Simulated values for dependent variables from the three models allow for an examination of the likely values of water quality enhancements and flood control/storm protection services in wetlands. The simulation also provides information on forecast variability relevant to validity assessments of the existing meta-analysis models.

The procedure in chapter 3 is applied to case studies of wetlands in Arrowwood National Wildlife Refuge (NWR), North Dakota; Blackwater NWR, Maryland; Okefenokee NWR, Georgia; and Sevilleta and Bosque del Apache NWRs, New Mexico. The Sevilleta and Bosque del Apache NWRs are modeled as a single unit because of their proximity to one another within a single ecoregion along the Rio Grande River. The choice of sites is intended to contrast major types of wetlands of the contiguous United States. Salinity, precipitation, temperature, and distance to ocean are examples of physical variations across these sites. Variability across the sites in income distribution, population density, and culture reasonably well represent the range of diversity that could be compared quickly within the scope of this analysis. All dollar values in this dissertation are adjusted for inflation to 2010 US dollars using the US Bureau of Labor Statistics CPI Inflation Calculator.

Chapter 4 discusses the process for constructing a novel meta-analysis database that focuses on benefit transfers for valuation of water quality provisioning and storm control/flood protection services associated with wetland ecosystems in the four case study NWRs. In this chapter, I analyze the novel meta-analysis database with an ordinary least squares multivariate regression.

Chapter 5 develops the novel parametric locally weighted least squares (PLWLS) estimator and compares results from the OLS analysis of the dataset as well as with the results from Chapter 3. Finally, Chapter 6 compares and discusses the validity and usefulness of the results obtained from analysis of the novel meta-analysis database with the OLS and PLWLS models and also with the results obtained from the previously published meta-analysis models.

CHAPTER 2

LITERATURE REVIEW

A wide body of scientific literature provides a basis for understanding the complex relationships between human communities and wetland ecosystems. In this chapter, I review some of the main contributions of scientific disciplines relevant to a qualitative and quantitative understanding of the relationship between wetlands and humans. I begin the literature review by briefly reviewing the historical patterns of wetland loss and wetlands ecology as it relates to human well-being. Modern spatial data on wetlands and the relevant classification schemes are also discussed. The role of federal protection of wetland in the United States, specifically as achieved through protected lands in the National Wildlife Refuge System (NWRS) is discussed next. Much of the discussion of the NWRS is focused on four case study refuges chosen for an initial assessment due to contrasting ecological and human population characteristics relevant to each site. The last two components of the literature review deal with the methods employed by economists to measure the benefits to people provided by wetlands. These economic methods can be divided into primary valuation methods and secondary valuation methods. The advancement of the scientific understanding of the analysis of primary valuation results for the purpose of valuation, known as meta-analysis benefit transfer is the focus of this paper.

A Conceptual Model of Ecosystem and Human Systems

Figure 1 is a conceptual diagram of the relationship between ecosystem structure and ecosystem service benefits to people derived from the work of Brown, Bergstrom, and Loomis (2007). The conceptual diagram represents a basic theoretical model of the linkages that give rise to ecosystem goods and services, associated with an impact on human welfare, with elements at the top of the figure supporting elements beneath. Ecosystem structure refers to the “biotic and abiotic components of an ecosystem (Brown, Bergstrom, and Loomis 2007, 334)," and the connections among these. Ecosystem functions and processes include the cycles and interactions among ecosystem structures that lead to the generation of ecosystem goods and services. Ecosystem goods and services impact human welfare and therefore can be understood in the context of economic value. Understanding these linkages is thought to be important for developing important structural models that allow for predicting the effects of management strategies that impact ecosystem structure or function on goods and services that impact human welfare.

The relationship between human well-being and ecosystems can be represented with mathematical functions. Let *E* represent ecosystem goods and services, let N represent ecosystem structure, and let the function *r()* represent the transformative effects on ecosystem structure known as ecosystem functions and processes. Equation (1) represents the creation of ecosystem goods and services that are potentially available for human consumption.

$$E=r(N) \tag{1}$$

The relationship in equation (1) must be modified in order to account for ecosystem goods and services that are produced with the intervention of firms. Because the focus of this dissertation is on ecosystem services that can be produced and consumed without the direct intervention of private firms, I do not develop the conceptual role of ecosystem goods and services as factor inputs in production of conventional market goods (Freeman 2003, 96).

An important qualification for modeling human-ecosystem interactions with secondary data includes the observation that meta-analysis benefit transfer models use reduced form equations (Smith and Pattanayak 2002). Woodward and Wui (2001), Brander, Florax, and Vermaat (2006), Ghermandi et al. (2010), and Brander, Brouwer, and Wagtendonk (2013) all model estimated ecosystem values directly as a function of ecosystem structure (i.e., wetland land cover) while controlling for variations in primary valuation method and socio-economic variables. Because reduced form, single equation models treat all variables as exogenous, more complex causal relationships are ignored, potentially introducing bias into estimates of causal effects, which is discussed more below. In general, scientists are not yet able to fully model with empirical data all of the conceptual relationships posited by the economic theory of ecosystem services.

Econometric Analysis and Causality

Economists and other empiricists have long searched for means of quantifying causal relationships. Econometric models have been developed that acknowledge the complex causal relationships among variables. With quantitative analysis of observational or non-experimental data, econometric models can be either structural or reduced form. A structural model is derived from a theoretical model and can be interpreted as a statement of causality (Goldberger 1972; Sims 1980). Much of the development of the econometric theory of identifying structural models has come from time series analysis where the time ordering of events is fundamental to causal arguments and definitions (Sargan 1958; Granger 1969; Engle, Hendry, and Richard 1983). This work has typically been in the context of macroeconomic analysis. In the time series context, the time ordering of observations leads to intuitive descriptions of more complex data generating processes relative to cross-sectional analysis and thus potentially more nuance with regard to the nature of exogeneity and endogeneity, such as notions of varying degrees of exogeneity (Engle, Hendry, and Richard 1983) and cointegrating relationships (Granger 1988). However, because the available data is cross-sectional, time series modeling techniques are only of limited use for explaining data generating processes.

The development of causal arguments using cross-sectional data has often focused on structural equation models that are estimated using an instrumental variables (IV) approach (Angrist, Imbens, and Rubin 1996). In addition to requiring a system of structural equations, the IV approach to estimation requires that certain variables be omitted from some structural equations. In any of the structural equations, the omitted variables allow for included variables to vary as a function of the omitted variables while the dependent variable is not directly affected, allowing for identification of the causal influence of included variables based on variations induced by the omitted variable.

A reduced form model often serves as an approximation to a structural model, and these models can be useful for situations when theory or data are insufficiently developed for estimating a structural model. While information about causality can be inferred with the use of a reduced form model, reduced form models are particularly useful for prediction exercises such as benefit transfer. All meta-analysis models in the ecosystem service literature can be interpreted as reduced form models, due to potentially endogenous explanatory variables, suggesting limited usefulness in testing hypotheses relating to causality. The models estimated below are reduced form models intended primarily for dependent variable forecasting, though there is no compelling reason to believe that the estimated parameters are less accurate estimates of causal effects than earlier meta-analysis models of wetland ecosystem services.

A closely related concept to the divergence between structural and reduced form models is the divergence between exogenous and endogenous explanatory variables. As changes in explanatory variables are intended to explain changes in the dependent variable, it is necessary that the changes in the explanatory variable are exogenous. The exogeneity of explanatory variables is a required assumption for estimating conventional regression models. The failure of this assumption can lead to bias and such failures are often an important critique of results obtained from regression models.

Goods and Services Beyond Conventional Markets

Economists have created several classification systems for defining and understanding both market and ecosystem goods and services. Of particular interest are those ecosystem services that are not found in conventional markets and which lack market prices. As ecosystem goods are tangible and values are more readily quantified, ecosystem services are the primary focus of this research.

The lack of market prices for ecosystem services is generally thought to have led to under-provisioning of these services. Two concepts have been used to explain the divergence of incentives that stakeholders face when making decisions that affect the availability of goods and services to themselves and other consumers: rivalry (also known as depletability) and excludability (Samuelson 1954). In order for a good or service to be priced in perfectly competitive markets, the owner of the good or service must be able to exclude non-payers from benefiting from using the good or service; without being able to do so, there is no incentive for a buyer to pay a non-zero price. In addition to the property of excludability, an efficiently priced good or service in a perfectly competitive economy will be sold at marginal cost. Rival good or services are depleted when one person enjoys them, making a unit of that good or service unavailable for others to enjoy. Due to the property of rivalry or depletability of most market goods, firms will provide these goods only at positive prices. Non-rival good or services can be supplied at zero marginal cost, as nothing must be done to supply an additional unit of the good or service when the existing supply has not been reduced by the prior unit of consumption. For those goods or services that have zero marginal cost of supply, there is no economically efficient positive price. This feature results in underprovision of beneficial non-rival goods and services to consumers by private, profit-maximizing firms.

Wetlands Literature Review

The structure of ecosystems and the services they can provide have changed dramatically in historical times. These changes and technological improvements and population growth resulted in a net loss of around half of all wetlands over the last 200 years (Gibbs 2000). As wetland ecosystems were less profitable and posed a number of nuisances to early farmers and nearby residents, the filling or draining of wetlands was for decades a widespread practice encouraged by government at various levels. Recently, interest in the lost capacity of wetland ecosystems to provide beneficial services such as water quality enhancements and flood control/storm protection has grown. Environmental economists have accordingly developed techniques to quantify the foregone benefits to humans due to wetland conversion as well as benefits enjoyed due to wetland conservation.

The attention that economists have paid to wetland ecosystems is by the nature of the study of economics an interest in how wetlands affect human welfare. The public nature of many wetland services precludes the use of conventional approaches to economic valuation due to the lack of market transactions for the preponderance of services provided by most wetland ecosystems. The understanding of wetland processes afforded by ecological studies and theory provide a useful basis for understanding in a qualitative sense how wetlands can potentially supply human populations with useful services as well as potentially deleterious disservices. However, identifying the interaction of human demand for ecosystem services with the supply of ecosystem services requires an economic approach. Generally, the pivotal role of water in wetlands indicates that wetlands are highly relevant to many water-related services. Consequently, much of the discussion of key ideas for ecosystem service valuation focuses on ecosystem services related to aquatic ecosystems.

An examination of the divergence between public and private goods and services associated with wetland ecosystems is useful for understanding the interactions between wetlands and surrounding human populations. Wetlands provide an abundance of ecosystem services that are public in the sense that they can be enjoyed by the public without payment (Heimlich et al. 1998). Initially, public policy focused on wetland conversion projects intended to increase the output of marketable goods. In the latter half of the 20th century, as stakeholders became more aware of the value of lost ecosystem service flows interest in and studies of the interactions between humans and wetland environments increased.

In general, wetland definitions in this dissertation follow The Convention on Wetlands of International Importance, referred to as the Ramsar Convention in defining wetlands as, “areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres” (Ramsar 1971, Article 1.1). This definition of wetlands is broad and inclusive of many water dominated landscapes, fresh and marine, including landscapes ranging from open water with rocky or sandy bottoms to densely vegetated swamps and marshes.

In practice, an important aspect of working with data related to wetland ecosystems is the classification and mapping of existing wetlands. Consequently, the functional definition of wetlands is implicitly derived from domestic wetland mapping programs. Several systems have been proposed for the classification of wetland ecosystems, and each is used in a prominent wetlands mapping system. Two prominent classification systems relevant to wetland ecosystems were described by Anderson et al. (1976) and Cowardin (1979) and were later implemented in the National Land Cover Database (NLCD) (Homer et al. 2007; Fry et al. 2008; Xian, Homer, and Fry 2009) and the National Wetlands Inventory (NWI) (USFWS 2012), respectively. The Anderson et al. (1976) system at level 1 distinguishes among open water, forested wetlands, and non-forested wetlands. The Cowardin (1979) hierarchal system employed by the NWI classified wetlands according to their system (Marine, Estuarine, Riverine, Lacustrine, and Palustrine), subsystem (e.g., subtidal, intertidal, lower perennial, upper perennial, intermittent, littoral, and limnetic), and class (e.g., aquatic bed, unconsolidated bottom, reef, rock bottom, rocky shore, emergent wetland, scrub-shrub wetland, forested wetland, and aquatic bed). Because NWI data were used in the development of NLCD data, the data sets largely agree on wetland extent.

The NLCD is useful because all lands of the contiguous U.S. are represented, including categories for developed lands, wetlands, and other uplands. Furthermore, the NLCD data sets are available for the years 1991, 2001, and 2006, providing a means for evaluating land cover changes, though differences in the 1991 analysis method may exacerbate inaccuracies in land cover change analysis. The NLCD 2006 data set is particularly useful because the included updated NLCD 2001 data were reassessed using updated techniques, increasing the accuracy of evaluations of land cover changes. The NWI data set is useful due to the high degree of categorical resolution afforded by the Cowardin (1979) hierarchal classification system. For example, the NLCD data sets do not distinguish between forested and scrub-shrub wetlands as the NWI data set does, and the NWI contains data useful for distinguishing among marine wetlands, estuaries, lakes, rivers, and swamps: the distinction between high and low salt content waters afforded by the NWI's fine categorical resolution is particularly useful for distinguishing among functionally different wetlands.

An important facet in the understanding of how human populations benefit from wetlands is the historical decline in wetland extent. Much of the impetus for wetland destruction is due to a demand for well-drained and fertile agricultural lands as well as a desire to reduce the populations of mosquitoes.

Early in US history, federal actions promoted the drainage of wetlands, but recently federal actions such as the Swampbuster Act have curtailed the conversion of wetlands in pursuit of a *no net loss* goal (Heimlich et al. 1998). Since the Farm Bills of 1985 and 1990, the Conservation Reserve Program and the Wetlands Reserve Program, respectively, have been instituted to reverse or prevent the destruction of the loss of important landcover types like wetlands that have poor agricultural productivity in their natural state. Generally, conservation efforts have tended to fall short of the goal of no net loss (Brown and Lant 1999).

Storm protection and flood control in addition to water quality enhancements are two important groups of ecosystem services associated with wetlands (Heimlich et al. 1998). In the context of agriculture, efforts made by individual stakeholders to enhance the drainage on their own land led to increased downstream flood risks. More systematic efforts made by the US Army Corps of Engineers, exemplified by the construction of levees, were implemented to forestall the increased risks associated with enhanced drainage of upstream agricultural soils. Ultimately, conventional investments in flood control infrastructure have failed to reduce flood damages, leading to renewed interest in the usefulness of wetlands for the storage of flood waters (Hey et al. 2004). Wetlands provide similarly useful temporary water storage services in the event of storm surges. The development of coastal resources away from their natural states along with trends towards rising sea levels has contributed to a decline in the quantity and quality of coastal wetlands, potentially exposing populations to greater damages from storms .

Wetlands serve a variety of important functions in the National Wildlife Refuge System. One of the fundamental goals of the NWRS is to provide habitat for various species, especially *trust species* (Fischman and Adamcik 2011). An important role of wetlands in the NWRS is the provisioning of habitat for economically important waterfowl, marine mammals, and other aquatic species. Protection of threatened and endangered species is also an important component of the NWRS mandate (Fischman and Adamcik 2011) and the established purposes for many refuges (Griffith et al. 2009). Wetlands are an important component of meetings this mandate due in part to their support for waterfowl populations.

Competing demands for refuge lands have been identified. Fischman (2002) describes tensions including those between the executive and legislative branches, between local demands for recreation and broader demands for conservation, and finally between the established purposes associated with individual land acquisitions at refuges and the broader goals of the NWRS. Generally, modern management of NWRs focuses on multiple uses with hierarchical priorities (Fischman 2002).

Public participation in refuge management decisions required by the NWRS Improvement Act of 1997 is considered by Fischman (2002) to fall short of good leadership. The use of ecosystem service values to inform management decisions is an example of developing forms of public participation that go beyond those mandated by law. In particular, the use of ecosystem service values for management decisions provides the US Fish and Wildlife Service (USFWS) a means of accommodating the tastes and preferences of people who do not participate in formal, legally required public review and comment (Fischman 2002).

In addition to supporting fulfillment of the congressionally mandated protection and conservation of the U.S.'s wildlife resources specified in the NWRS Improvement Act of 1997, wetlands also provide important water-related benefits both on and off site. In order to protect these benefits, federal law allows and at times requires refuges to secure water rights in a manner that supersedes state laws in order to fulfill refuge establishment purposes as well as NWRS mandates. Limited funding has been identified as a barrier to attaining these goals (Fischman 2002).

There are few existing primary valuation studies of any National Wildlife Refuge and fewer that focus on wetlands. Examples of published value estimates for NWRs obtained with secondary data are limited to the work of Ingraham and Foster (2008). Several closely related primary valuation studies focusing on wetland resources in NWRs have been conducted in California's Central Valley (Hanemann, Loomis, and Kanninen 1991; Loomis et al. 1991; Park, Loomis, and Creel 1991; Creel and Loomis 1992). Klocek (2004) is an example of a NWR valuation study that does not allow one to isolate a value for wetlands from the refuge in general.

This dissertation contributes to the understanding of the role of NWR wetlands in providing the public with beneficial ecosystem services through the measurement of the economic values of water quality enhancements and flood control/storm protection. The next section discusses the economic theory of measuring these benefits to domestic populations.

Economic Theory and Welfare Measures

Neoclassical economists posit the existence of a utility function that can be used for considering changes in an individual's welfare and for assessing trade-offs between various goods and services. Ecosystem service valuation is fundamentally a concern of how individuals are affected by ecosystems, and these affects are measured in units of money to allow comparability with other sources of well-being.

The foundation of the economic approach to quantifying welfare changes in individuals, i.e., the microeconomic approach, is the assumption that consumers are rational. Rationality consists of more primitive assumptions or preference axioms. Wetzstein (2005) provides a suitable definition of preference axioms. For the representative consumer to be rational, Wetzstein (2005) assumes that the consumer has a complete ordering of preferences over all possible consumption bundles and that those preferences follow the rules of reflexivity and of transitivity. In order to claim that the representative consumer maximizes a utility function, Wetzstein also assumes that preferences are continuous, that utility is monotonically increasing with consumption, that the consumer is non-satiable, and that indifference curves are convex.

In addition to conventional market goods, non-market public goods also affect a representative consumer's utility. Public goods have two features that prevent them from being priced efficiently in perfectly competitive markets, non-depletability (or non-rivalry) and non-excludability (Freeman 2003, 3). Non-depletability means that one person's enjoyment of a public good does not reduce the amount available for another's enjoyment (Samuelson 1954). Non-excludability refers to constraints that prevent an owner of a good from excluding non-payers from enjoyment of the good. A consequence of non-depletability is that the marginal cost to produce an additional unit of the good for consumption is zero, implying that the price of the good will be zero in a perfectly competitive market. The consequence of non-excludability is that a consumer, lacking an incentive to do so, is not willing to pay a positive price to enjoy the good, as non-payment and payment do not affect accessibility. Accordingly, theory supports the expectation that public goods are under-provided by private firms relative to the magnitude of consumer demand and the costs of producing those goods.

I roughly follow Freeman (2003) in conceptualizing the role of private market and public ecosystem goods and services in the utility of consumers. In order to include public goods in the economic analysis, I assume that a representative consumer's utility is a function of both private goods and services, X, and ecosystem services that are public goods, E. Generally, services are defined in such a manner that their marginal utility is positive for the level of consumption relevant to the analysis. This definition of E implies that $\partial U(X,E)/\partial E>0$ in the utility function or equation (2).

$$U=U(X,E) \tag{2}$$

Because private goods and services must be purchased in order for them to affect utility, income, m, and prices, P, are the arguments of the budget constraint of the consumer's utility maximization problem,

$$X'P=m, \tag{3}$$

where X and P are column vectors of quantities of market goods and their respective prices. Maximization of (2) subject to the constraint in (3) leads to the Marshallian demand functions for the market goods. Substituting these demand functions into the utility function leads to the indirect utility function, $V(.)$, where

$$v=U(X(P,m,E),E)=V(P,m,E). \tag{4}$$

In the middle term of equation (4), the indirect utility function might also explicitly be defined such that E is a function of market prices and income due to complementarity that may exist between certain ecosystem services and market goods. The indirect utility function, when appropriately specified, can be solved for income, m, which by duality results in the expenditure function, $e(.)$.

$$e(v,P,E)=m \tag{5}$$

In equation (5) v represents a specific level of indirect utility that can be satisfied at prices, P, and with available ecosystem goods and services, E, that requires a total expenditure on market goods and services equivalent to m. The expenditure function is useful for defining measures for quantifying changes in welfare.

There are two commonly used groups of welfare measures in neo-classical economics, Hicksian and Marshallian. Measures of Hicksian consumer surplus or compensation are useful for answering questions about trade-offs between ecosystem service flows and monetary compensation associated with changes in ecosystem goods and services, which is the primary interest in this study. Stated preference approaches are particularly apt for measuring Hicksian surplus or compensation as the respondent is asked the compensation question through choice or via direct elicitation. Marshallian consumer surplus is more appropriate for quantifying the net benefits associated with market purchases, but this measure is also bounded by Hicksian welfare measures for willingness to pay (WTP) and willingness to accept (WTA) and it has been argued that Marshallian consumer surplus can be reasonably used as an approximation of the more theoretically desirable Hicksian welfare measure (Willig 1976). Randall and Stoll (1980) developed the theory for better understanding the relationship between the welfare measures for quantity changes that are often relevant for wetland ecosystem services.

Hicksian measures of a consumer's surplus due to changes in the availability of ecosystem goods and services, E, can be readily demonstrated with the expenditure function in equation (5). Typically, the change in E is assumed to have no impact on prices, P, of market goods, but the expenditure function implicitly allows for optimizing changes in the quantities of market goods and services purchased. For a valuation scenario where the initial level of utility (that is, prior to a change in E) is of interest, Hicksian compensating surplus is the appropriate welfare measure. Hicksian compensating surplus, c, is defined according to the following:

$$\begin{gathered} \mathrm{m} = e(v,P,E) = e(v,P,E') + c \\ c = e(v,P,E) - e(v,P,E') \end{gathered} \tag{6}$$

where *E'* denotes a changed availability of ecosystem goods and services from the original availability, *E*. The variable *v* indicates the initial level of utility prior to the change in *E*.

Measures and approximations of Hicksian surplus are useful for public goods such as ecosystem services which are readily conceived of in terms of measures of quantity because they can be partially or entirely indivisible and the quantity consumed cannot be altered by a consumers as often occurs with price changes for market goods (Randall and Stoll 1980). While summing estimates of Hicksian compensation over multiple people is not without contention (Boardway 1974), this practice is essentially the only choice for evaluating whether a policy leads to a potential Pareto improvement. The next section discusses methods for measuring the economic values of ecosystem services.

Primary Valuation Techniques Literature Review

The theoretical foundation for a wide variety of non-market valuation techniques lies in the random utility modeling (RUM) approach to experimental design and statistical modeling. Thurstone (1927) is typically credited with the original idea behind RUM (e.g., Holmes and Adamowicz 2003), while the statistical implementation in the context of a utility maximizing consumer via discrete choice experiments was developed by McFadden (1973). The RUM approach to modeling is especially useful in the study of ecosystem services due to the connection with formal economic theory and consistency with the notion of a rational, utility maximizing individual. Both stated preference and revealed preference studies can be designed by researchers to yield data that can be modeled with the RUM framework.

Stated Preference Valuation Methods

The basic approach with stated preference studies is to ask people how much compensation they are willing to pay (WTP) in order to avoid (or obtain) a negative (or positive) change in ecosystem service flows. Often, such as in cases where consumers were illegally deprived of valuable ecosystem services, willingness to accept (WTA) is the more appropriate concept, but for a variety of reasons, in empirical applications willingness to pay is used instead. Generally a consumer's rights determine whether WTP or WTA is appropriate (Carson, Flores, and Meade 2001).

Economists define the response to a stated preference valuation question in terms of the expenditure function, as seen in equation (6). In equation (6), two states of reality are depicted on each side of the equals sign. The left hand side represents a consumer's expenditures as a function of a non-changing level of utility, *v*, an unchanging price level in the market economy, *p*, and changing availability of ecosystem services, from *ess* to *ess'*, associated with a level of compensation, *c*. If the ecosystem service flows represented by *ess'* are valued less by the consumer, then compensation, (*c*), will be positive, offsetting the need for greater expenditures on market goods in order for utility to be unaffected. In this conceptualization of the value of ecosystem services, the welfare concept is known as Hicksian compensating surplus or Hicksian equivalent surplus; the former implies that a consumer has the right to the level of utility, *v*, prior to a change, while the latter implies a consumer's right to the level of utility (*v'*) after a change.

The Contingent Valuation Method

The contingent valuation method (CVM) is the most straightforward approach to eliciting Hicksian compensating or equivalent surplus values via stated preferences in order to quantify changes in human welfare. Carson and Hanemann (2005, 827) attribute the development of the approach for valuing goods and services lacking market prices by Bowen (1943) and Ciriacy-Wantrup (1947), and the first criticism is attributed to Samuelson (1954), who discussed the potential for strategic responses. The method gained traction in the 1980's (e.g., Mitchell and Carson 1989), and generated a good deal of controversy in the 1990's and early 2000's. With CVM a consumer may be asked directly how much she is willing to pay (open-ended) or alternatively whether she would be willing to pay a certain amount (dichotomous choice) and in some cases the process may be iterated (Boyle, Bishop, and Welsh 1985; Bateman et al. 1995). One particular strength of CVM is the ability to value ecosystem service flows that are hypothetical, which is generally not a possibility with revealed preference approaches (Carson, Flores, and Meade 2001).

While a number of important criticisms of the CVM approach to valuation exist, the approach was found by a National Oceanographic and Atmospheric Administration (NOAA) scientific panel (Arrow et al. 1993) to be fundamentally (i.e., when best practices are followed) reliable for evaluating welfare changes in situations where alternative valuation approaches fail to fully measure the values in question due to omission of passive-use values. The panel found that, in particular, *Burden of Proof* requirements must be met for the results to be reliable for their use in natural resource damage assessments which are then legally valid for determining compensation for damages. Despite the endorsement of the CVM method by prominent Nobel Laureates on the NOAA CVM panel, a number of subsequent studies sharply criticized the reasoning of the panelists and ultimately the usefulness of information from CVM surveys; these and some rebuttals are discussed below.

Several overlapping criticisms of the CVM were published around and following the 1993 NOAA panel released its recommendations. These criticisms came in the context of the Exxon Valdez oil spill and the incipient legal framework for protecting public resources with potentially large and difficult to quantify existence values (Portney 1994). For example, the *scope test* has received considerable attention in the ensuing literature. The *scope test* essentially requires that WTP varies in a manner that is weakly monotonic with changes in the quantity or quality of the ecosystem service flow associated with the valuation question. Kahneman and Knetsch (1992) along with Diamond and Hausman (1994) express strong skepticism about the usefulness of CVM responses due to *embedding effects*, which occur when an individual expresses a different value when a valuation question is embedded in a different place in the questionnaire such as when valuation questions are ordered differently (Carson 1997a). Embedding effects in particular have been associated in these criticisms with failures of the scope test. The CVM is also criticized as being fundamentally vulnerable to problems from *hypothetical bias*, which results from a divergence between what people might be observed doing and what they say they would do (Cummings, Ganderton, and McGuckin 1994; Cummings, Harrison, and Rutström 1995).

A number of more recent examinations have concluded that CVM is valid. Carson, Flores, and Meade (2001) attribute to the criticisms the emergence of, "a much richer theoretical framework for non-market valuation…." The same authors suggest that the questionable results that spurred criticisms of CVM are indicative of problems with the information in the survey rather than the broader approach (Carson, Flores, and Meade 2001).

The criticism that CVM too often fails the scope test has also been rejected by a number of authors. Carson, Flores, and Meade (2001) criticize the conclusions about CVM made by several authors (e.g., Kahneman and Knetsch 1992; Diamond and Hausman 1994) based on their observation that a review of empirical studies find that CVM results do in fact pass the scope test, both within a single sample and across multiple independent samples. Carson, Flores, and Meade (2001) further suggest that a study with results failing the scope test is likely indicative of quality problems in that study. The related embedding effect that often leads to failures of the scope test has been similarly rejected as a fundamental vulnerability of CVM studies. Carson (1997b) reviews studies (Hoehn and Randall 1989; Hanemann 1991) that justify what appears to be embedding effects by a careful analysis of economic theory, concluding that these effects do not threaten the validity of CVM.

The Choice Experiment Method

Choice experiments or choice modeling are a stated preference method that is considered by some to be a type of or variant of CVM (Adamowicz et al. 1998). Choice modeling studies are also referred to as a*ttribute-based methods* by some environmental economists (e.g., Holmes and Adamowicz 2003). Typically respondents are presented with a series of choice occasions. A choice occasion includes two or more options for a respondent to choose from with variability introduced through attributes of the choice; for valuation results a price or cost to the respondent is necessary and also serves as a source of variability across choices. Choice experiments are particularly useful for highlighting for the respondent the attributes that are being varied (Adamowicz et al. 1998).

Choice modeling experiments typically utilize the random utility modeling framework developed by McFadden (1973). As discussed above, this feature of choice experiments allows for the estimation of theory based measures of welfare changes. Because choice modeling experiments require stated preference responses to survey questions, most if not all critiques and rebuttals of CVM also apply. Generally, choice experiments are thought to be most applicable to valuing attributes that may be changed by a management action while CVM is able to value the entire system (Hanley, Wright, and Adamowicz 1998). Hanley et al. (1998) also consider choice experiments less vulnerable to "yea-saying", as individuals cannot simply agree as they might in a dichotomous choice CVM experiment. Nonetheless, ordering effects of choices may lead to a similar problem with responses, though the approach can be designed to test for embedding effects (e.g., Johnston et al. 2002). Also, as a choice experiment often involves a number of choice occasions, it is possible to examine an individual's responses to test for internal consistency (Bush, Colombo, and Hanley 2009).

As with other stated preference approaches, ensuring that individuals understand the valuation question is paramount to obtaining theoretically correct values. Because choice experiments require respondents to evaluate a number of options with several varying attributes, the burden on respondents of understanding the choices available are greater than with a single scenario presented in a typical CVM study. The use of visual supplements to the choice question have become a common remedy for anticipated difficulties a respondent may face with evaluating the implications of a potential choice (Bateman et al. 2009). Recently Bateman et al. (2009) address concerns that the tabulated attributes presented on each choice occasion are not easily evaluated by respondents. Based on a split sample approach they conclude that virtual reality presentations of the choices lead to a smaller difference between WTP and WTA measures of compensation and less variability in responses in general.

Revealed Preference Valuation Methods

The Travel Cost Method

The travel cost method, according to Mitchell and Carson (1989), was first used to value outdoor recreation by Marion Clawson in 1959 (Clawson 1959). The travel cost method is typically based on an estimation framework of a household production function with weak complementarity between a non-market good or service and a market good or service (Freeman 2003, chap. 4). Typically, travel cost studies have estimated Marshallian consumer surplus as the Hicksian demand curves and expenditure function are not readily observable, whereas the Marshallian demand curve is observable.

As Hicksian welfare measures are often most relevant to answering public policy questions that relate to changing ecosystem service flows, questions have arisen about the usefulness of Marshallian consumer surplus estimates for answering these questions. Willig (1976) cites two authors in particular that discount the usefulness of Marshallian consumer surplus as an approximation of Hicksian measures of compensation. Willig's study was the first to quantify the gap between the groups of welfare measures. This development established a theoretical means for evaluating the magnitude of error associated with using Marshallian welfare measures in place of more theoretically appropriate Hicksian welfare measures. Generally, the error associated with using the Marshallian welfare measure as an approximation of the Hicksian welfare measure is only small for small changes that do not shift the demand curve (Freeman 2003, 423). Welfare changes that are based on shifts in the Marshallian demand curve, which can occur when measuring the value of changes in site attributes, cannot be analyzed with Willig's method (Freeman 2003, 427).

Modern modeling approaches often utilize the RUM framework to allow for researchers to value sites in the context of recreation participants facing multiple site choices. Such discrete choice methods allow for identification of the welfare effects associated with various attributes at each site on the probability of an individual visiting that site and can be used to estimate the value of that site.

The Hedonic Price Method

The hedonic price method (HPM) has been used for valuing the contribution of ecosystem services to human welfare since Ridker and Henning's (1967) work on the effects of air pollution on residential property values. Rosen's (1974) formalization of the theory of hedonic pricing in the context of utility maximization further spurred applied work in this area. Hedonic price method studies estimate implicit prices of housing attributes with a first stage regression and compensated and uncompensated demand curves associated with those housing attributes can be derived and estimated with a second stage regression (Rosen 1974; Taylor 2003, 364).

A number of meta-analysis models have focused on the hedonic price method (Smith and Huang 1995) or included hedonic price method primary valuation studies (Woodward and Wui 2001; Brander, Florax, and Vermaat 2006; Ghermandi et al. 2010; Brander, Brouwer, and Wagtendonk 2013). Hedonic valuation studies quantify implicit prices and sometimes consumer surplus for services that are purchased through the housing market, making these services somewhat different from services modeled with TCM

Other Measures of Welfare Changes

A variety of methods based on market prices have been developed to quantify the impact of changing ecosystem service flows on human welfare. Typically these methods require somewhat stronger assumptions in order for results to be treated as economic measures of willingness to pay; alternatively these methods may be useful for establishing bounds on willingness to pay. A number of primary valuation studies that appear in published meta-analysis studies of ecosystem service values use one of the methods below.

Cost Based Methods: Damage Avoidance, Averting Expenditures, and Replacement Cost Method

Several approaches are available to quantify welfare impacts of ecosystem services based on the idea of damage avoidance. Damage avoidance methods estimate the damages that would have occurred in a different ecosystem or landcover configuration and assumes that the value of these avoided losses is an approximation of the relevant population's willingness to pay for the change in the ecosystem. Costanza and Farber (1985) and later with more sophisticated methods Costanza et al. (2008) both use a damage avoidance approach to attribute changes in hurricane damages to changes in wetland acreage, for example. Fundamentally, this approach assumes that structures and other capital damaged by storms is worth to the local population what it would cost to replace or what it originally cost to build. Changes in the relevant population's tastes and preferences may make this assumption faulty; results may be biased in either direction.

The alternative conceptualization of a damage avoidance study focuses on expenditures made to avoid future damages. These expenditures are known as averting or defensive expenditures and are often studied in the context of pollution (Courant and Porter 1981; Abdalla, Roach, and Epp 1992). Averting expenditures are also often classified as a type of revealed preference approach to quantifying household welfare when averting expenditures are made at the household level (Brown, Bergstrom, and Loomis 2007). While water quality and flood control/storm protection welfare impacts can conceivably be approximated by the averting expenditures approach, I am unaware of studies that have done so for domestic US wetlands.

The replacement cost method focuses on the costs of replacing ecosystem services with built capital that can perform the same service. Replacement costs estimates can be overestimates of the welfare effect of the service if the replacement cost is higher than a population is willing to pay. In some instances, replacement would be legally mandated, such as when failing to do so would lead to violations of the clean water act; such mandates increase the validity of replacement cost based welfare estimates, but skepticism of the approach remains (Brown, Bergstrom, and Loomis 2007). Replacement costs studies have been used to quantify the value of wetlands for waste water processing (Fritz, Helle, and Ordway 1984; Breaux, Farber, and Day 1995).

Production Function and Market Price Techniques

Many ecosystem goods or services are associated with the production of a final product that is purchased in a conventional market. Production functions can be specified and analyzed in order to develop theoretically accurate measures of the contribution of ecosystems to human welfare through inputs to useful production processes (Brown, Bergstrom, and Loomis 2007). Production functions may focus on firms that employ built capital to produce in a conventional manner for firms as well as on the ecosystems themselves which can be thought of as producing useful outputs such as fish available for recreational fishing (e.g., Batie and Wilson 1978; Bell 1997) or commercial fishing (Ellis and Fisher 1987; Lynne, Conroy, and Prochaska 1981; Costanza, Farber, and Maxwell 1989). The novel meta-analysis in this study does not employ production function methods, as market prices can be used along with models of ecosystem productivity to estimate how benefits from market exchanges relate to wetland management.

Secondary Data Valuation Techniques Literature Review

Benefit Transfer Literature Review

Benefit transfer techniques were originally developed to gain an understanding of the recreation benefits that might be supported by hypothetical reservoirs should they be built (Loomis 1992). Prior to the development of meta-analysis approaches, function and unit transfers were the main options available to researchers. The earliest benefit transfers were limited to expert judgment (Walsh, Johnson, and McKean 1992), a method that is no longer commonly classified as benefit transfer (Johnston and Rosenberger 2010). I follow much of the existing literature in referring to the site associated with a primary valuation study estimate as a study site, and I generally refer to an unstudied site that receives a value transfer as a policy site (Brouwer and Spaninks 1999).

Generally, benefit transfers are expected to perform better when more information is used. For example, the additional information contained in benefit transfer models that incorporate multiple primary valuation studies can allow researchers to control for variations in benefits induced by differences in landscape, socio-economic, and method related variables.

Several approaches to benefit transfer with existing ecosystem service valuation data have been developed. Aside from expert judgment as a benefit transfer tool (Johnston and Rosenberger 2010), benefit point or unit transfers are the most basic and utilize a single value from a primary valuation study outside of the context of the original valuation estimate. Costanza et al.'s (1997) global valuation work is one of the most well-known examples of point transfer, with a variety of point transfers used for different land cover types across the world's landscapes. A recent study applied a similar approach to the state of New Jersey (Costanza et al. 2007; Liu et al. 2010). Bockstael and McConnell (2007) review studies that voice a number of concerns about the appropriateness of the point transfers employed in the Costanza et al. (1997) study including concerns about how multiple benefit estimates are aggregated and concerns about valuing all ecosystem services at once using economic concepts developed for small or marginal changes. Bockstael et al. (2000) criticize the Costanza et al. (1997) study and provide a non-technical overview of ecosystem service valuation, including the problems associated with valuation questions associated with very large, non-marginal changes in ecosystems and poorly defined valuation questions that lack practical policy applications. The first issue of volume 25 of the journal, Ecological Economics contains a number of technical criticisms of the Costanza et al. (1997) valuation study.

Function transfers are the primary alternative to point or unit transfers, and are often considered to be more accurate on average (Rosenberger and Stanley 2006). Early function transfers used information beyond a welfare estimate, such as transferring demand equations for valuing recreation sites (e.g., Loomis 1992). While Johnston and Rosenberger (2010) classify meta-analysis benefit transfers as a type of function transfer, most meta-analysis studies only transfer point estimates, but estimate a benefit transfer function in order to incorporate multiple point estimates into a single transfer.

Questions pertaining to the impact on transfer accuracy of the degree of site and study similarity between study sites and policy sites have arisen since the early days of benefit transfer (e.g., Boyle and Bergstrom 1992). Johnston (2007) refers to this idea as the “similarity hypothesis”. A number of publications refer to this same notion as correspondence (Rosenberger and Stanley 2006; Rosenberger and Phipps 2007; Rosenberger and Johnston 2009; Johnston and Rosenberger 2010). Rosenberger and Johnston (2010) review the main issues associated with benefit transfers, with a substantial focus on issues that relate to transfer error.

Meta-Analysis Literature Review

The emergence of quantitative meta-analysis is often attributed to the educational psychology researcher, Gene V. Glass (i.e., Glass 1976). The earliest meta-analysis studies relating to ecosystem services focused on recreation demand, with Smith and Kaoru (1990a) examining demand elasticities for recreation and Smith and Kaoru (1990b) examining consumer surplus from travel cost recreation studies.

In my search of the literature on wetland ecosystem service valuation studies, I used references and supplementary data from several existing meta-analysis studies of wetland ecosystem services to track down relevant primary valuation studies. Table 1 contains a summary of the existing wetland meta-analysis studies identified in the literature search. As pointed out by Moeltner and Woodward (2009), most meta-analysis studies are broad and intended more to summarize the literature and for hypothesis testing. Moeltner and Woodward’s (2009) meta-analysis study is the only wetland specific meta-analysis designed for a specific benefit transfer application I was able to find in the literature.

A number of studies (e.g., Hanley, Wright, and Alvarez-Farizo 2006; Johnston and Thomassin 2010) conduct a meta-analysis of WTP estimates for environmental improvements. While their reference lists are useful for tracking down potentially useful primary valuation studies (those which attribute estimated values to specific wetland ecosystems) the dependent variables in these studies require additional information in order to relate predicted values to wetland ecosystems without a supplementary ecological model to indicate the wetland extent required to achieve a specific improvement. Similarly, the meta-analysis of Moeltner, Boyle, and Patterson (2007) considers recreational fishing values that are WTP/person/day, requiring a conversion such as one that estimates fishing days per acre of wetlands. The included meta-analysis models that normalize WTP by household or individual (Brouwer et al. 1999; Moeltner and Woodward 2009) focus on specific wetland landscapes where acreage can be quantified, which allows for reasonably straightforward conversion to WTP per person per acre of wetland.

All of the broad meta-analysis studies use a similar regression equation specification, where the dependent variable is estimated annual willingness to pay normalized by surface area or a population count. The lone exception is Borisova-Kidder's (2006) dissertation, which models WTP as WTP per person per acre. In these studies, the independent variable groups include site descriptor variables, variables related to the surrounding geographic and socio-economic conditions, method-related attributes of the primary study, dummy variables for the ecosystem service valued, and dummy variables for the quality of the primary study (only found in Woodward and Wui 2001). The regression error term is assumed to be an independently distributed, mean zero stochastic variable.

The broad wetland ecosystem service meta-analysis studies tend to introduce incremental improvements in method and incremental expansions of the sample size based on comparable sampling mechanisms. For example, improving on Woodward and Wui (2001), Brander, Florax, and Vermaat (2006) include socio-economic and geographic variables to control for variations in the underlying landscape structure and the user population. Ghermandi et al. (2010) also include variables to control for man-made wetlands and regional substitutes (area of wetlands in a 50km radius around the center of each site). Additionally, Ghermandi et al. (2010) consider weighted regressions in response to increased understanding of the need to account for unobserved similarities among observations from a single study (Nelson and Kennedy 2009). Finding little impact on the results, they retain the unweighted model. The wetland meta-analysis studies of Woodward and Wui (2001), Brander, Florax, and Vermaat (2006), and Ghermandi et al. (2010) form a sequence of studies with a similar broad focus on a variety of services and wetland types; this literature review focuses on these three studies.

Despite differences in the explanatory variables included in the studies, all three in the identified sequence of broad similarly focused wetland meta-analysis studies find significant and positive but decreasing marginal returns from additional wetlands. In these three studies, comparisons of the estimated effects of wetland type across meta-analysis studies are hindered by the use of different wetland classification schemes. The greatest similarity across meta-analysis studies is the inclusion of dummy variables to control for the effect of the type of service valued. Water quality provisioning is of relatively high value in the three similar meta-analysis studies, while flood control is valuable to a slightly lesser extent. The fit of the broad meta-analysis models are relatively comparable with each explaining roughly half of the variation in the dependent variable.

Meta-Analysis for Hypothesis Testing vs. Meta-Analysis for Benefit Transfer

Smith and Pattanayak (2002) described at least three purposes for meta-analysis of ecosystem services: research synthesis, hypothesis testing, and benefit transfer. It was later recognized by Moeltner, Boyle, and Patterson (2007) that the literature contains meta-analysis models used for two basic purposes, hypothesis testing and benefit transfer. Hypothesis testing is typically based on inference obtained with estimated means and variances of regression parameters (e.g., Brander, Florax, and Vermaat 2006).

Benefit transfer studies such as that of Moeltner, Boyle, and Patterson (2007) use a rule-based approach to resampling from available studies in order to achieve high correspondence between characteristics of observations in the sample and characteristics associated with the study site and population. In their literature review on this topic, Moeltner and Woodward (2009) identify only a single other study (Smith and Huang 1995) in the academic literature that conducts an original meta-analysis for benefit transfer, which focuses on air quality.

Review of Best Practices and Pitfalls

Generalization error is a source of error in meta-regression estimation and meta-analysis benefit transfer. Rosenberger and Stanley (2006) describe generalization error as occurring when, “a measure of value is generalized to unstudied sites or resources”, and go on to hypothesize that this error will correlate inversely with site correspondence.

The use of a sample of primary valuation studies that include estimates of Marshallian and Hicksian welfare measures has been questioned due to concerns about situations in which these measures diverge (Smith and Pattanayak 2002). The potential for close proximity of various Marshallian and Hicksian measures to each other (Willig 1976) and the multiple sources of modeling error (Randall and Stoll 1980), and the conceptual flexibility during valuation of treating landscape changes as either price or quantity changes (Boyd and Banzhaf 2007), all suggest that in certain situations meta-analysts may relax standards implied by the concept of *welfare measure consistency*. *Welfare measure consistency* is discussed in Smith and Pattanayak (2002) and Bergstrom and Taylor (2006) and again in the review paper of Nelson and Kennedy (2009); this concept requires that the studies comprising the meta-data in the original meta-analysis have the same welfare measure. *Welfare measure consistency* is not present in the broad meta-analysis models of Woodward and Wui (2001); Brander, Florax, and Vermaat (2006); or Ghermandi et al. (2010).

Economic theory offers little guidance with respect to the appropriate treatment of method-related variables for meta-analysis benefit transfer estimation. While including dummy variables to control for the potential influence of primary valuation method on estimated values is a standard practice when applicable, the choice of how to code the method-related explanatory variables for prediction is thought to be poorly guided by economic theory. For example, a popular, ad-hoc choice for meta-analysis benefit transfers is to set method-related variables to their sample means (Rosenberger and Johnston 2009; Stapler and Johnston 2009). Stapler and Johnston (2009) find this treatment to be a close approximation to setting method-related variables to their known values for sites where primary valuation studies exist.

Economic theory predicts that stated preference and revealed preference studies will produce consistently different estimates of willingness to pay due to the different aspects of ecosystem services these methods value. However, the hypothesis of convergent validity across primary valuation estimators that measure the same set of services has found mixed support. Woodward and Wui (2001), for example find wide variation in the partial effects across revealed preference methods. Brander, Florax, and Vermaat's (2006) findings support convergent validity with significantly different parameter estimates only between revealed and stated preference estimated parameters. Ultimately, in order to produce estimated values that can be validated with primary valuation studies, I implement benefit transfer by coding for a specific method, and interpreting the estimated results as a forecast of the results should the selected method be implemented at the policy site.

The *n versus k* dilemma, discussed by Moeltner, Boyle, and Patterson (2007) provides a useful framework for discussing the problems faced by a meta-analysis practitioner in the course of data acquisition and model specification. In a situation where the practitioner has in hand an ideal data set with n observations, each containing measurements of the dependent variable and k identified explanatory variables, estimation of a conventional OLS model is straightforward. The dilemma arises due to certain observations lacking sufficient information on all variables. Obtaining all explanatory variables for all observations is usually not possible without compromises. Typically meta-analysts simply cannot use certain studies due to insufficient information contained in publications related to the study. Moeltner, Boyle, and Patterson (2007) provide one of the few alternatives to entirely discarding studies with insufficient information to code all regressors. Their approach is to use these studies to form prior expectations in a Bayesian estimation framework.

Sample Selection

Recent papers discussing the theoretical challenges of meta-analysis have pointed out the potential for inconsistent regression results due to sample selection bias (e.g. (Rosenberger and Johnston 2009; Rosenberger and Stanley 2006; Rosenberger and Phipps 2007; Bergstrom and Taylor 2006; Smith and Pattanayak 2002; Hoehn 2006). Several sources of selection bias have been proposed in the literature, such as research priority selection (Hoehn 2006) and publication selection bias (Rosenberger and Stanley 2006). Several efforts to address selection bias have focused on the Heckman correction, including an estimate of the Inverse Mills Ratio derived from a first stage regression as an intercept shifting explanatory variable in the meta-regression equation. In the meta-analysis of Hoehn (2006), for example, the Heckman model is used to control for research priority selection to address concerns of bias and inconsistency of parameter estimates for variables correlated with factors affecting research priority. The coefficient on the inverse mills ratio parameter in their second stage panel data model is significant at the 95% level in one specification considered and at the 90% level in another.

Rosenberger and Johnston (2009) describe a variety of sources as well as tactics for mitigating sample selection effects found in the meta-analysis benefit transfer literature. For example, a simple way to model selection assumes that selection effects can be modeled by inclusion of dummy variables describing the avenue of publication, i.e. working paper, journal article, dissertation (e.g., Woodward and Wui 2001; Brander, Florax, and Vermaat 2006). Publication selection bias, a prominent source of selection related error, occurs due to a non-random selection effect on potential non-market valuation studies, typically attributed to a greater demand by journals for method-related contributions and statistically significant results, as well as a desire for "results that conform to theoretical expectations (Rosenberger and Stanley 2006)." Another source of selection pressure discussed by Rosenberger and Johnston (2009) relates to the difficulty of finding values for all desirable variables to include in the regression, known as the N versus K dilemma, which may systematically drive researchers to exclude certain studies from meta-analysis data sets.

Loess Regression Literature Review

Locally weighted regression procedures offer analysts an alternative to the conventional single equation approach to regression. The first robust local regression estimators were developed by Cleveland (1979). Known as Loess or Lowess regressions, the method differs from conventional full-sample regression by relying more heavily on observations in a particular data point's neighborhood when modeling the behavior at that data point, typically with a polynomial specification (Cleveland 1979;

Cleveland and Devlin 1988). The use of such a method relaxes the assumption that the effects of independent variables are homogeneous across the sample.

An important aspect of Loess models concerns how a relationship between observations is defined as being local or not. The use of a smoothing parameter supplied by the analyst is typical. The specification of a value for this variable is subjective, requiring the analyst to balance concerns about incorporating too few observations and capturing random error versus concerns about capturing too many observations and failing to model the local pattern appropriately. Other analyst decisions are also required, such as specifying a functional form for the weighting function; which is somewhat analogous to the task in single equation regression procedures of specifying a functional form of that equation.

Valuation Linkages	Examples	
Ecosystem Structure	Wetland Vegetation, Water Quantity and Quality, Nutrient Levels, Fish and Wildlife Populations	Ecology
Ecosystem Functions/Processes	Primary Production, Secondary Production, Food Chain/Web, Nutrient Cycling, Hydrologic Cycle (surface and ground water flows)	
Ecosystem Goods and Services	Recreational Fishing and Hunting (days and catch), Wildlife Observation, Carbon Sequestration, Flood Control	Economics
Ecosystem Values	WTP for Fishing or Hunting Day WTP for Wildlife Observation Day WTP for Carbon Sequestration Flood Damage Avoidance	

Figure 1: Conceptual Relationships and Examples

Table 1: Previous meta-analysis studies of wetland ecosystem services.

Authors	# of Studies, # of observations	WTP/year normalization	Variable groups	Broad or specific	Other
Brouwer 1999	30, 92	per person	service, landscape, method,	broad	CVM only, GLS-multi-level model
Woodward and Wui 2001	39, 65	per acre	service, landscape, method, quality	broad	OLS
Brander, Forax, and Vermaat 2006	80, 215	per hectare	socio-econ, service, landscape, method, substitutes	broad	OLS
Borisova-Kidder 2006	33, 72	per acre per household	socio-econ, service, landscape, method	broad	OLS, Woodward and Wui (2001) database + 7 studies
Moeltner and Woodward 2009	9, 12	per household	socio-econ, landscape, user count	specific	Bayesian, benefit transfer focus
Ghermandi et al. 2010	170, 418	per hectare	service, landscape, method, artificial wetlands, substitutes	broad	OLS, WLS
Brander, Brouwer, and Wagtendonk 2013	38, 66	per hectare	Service, landscape, method, artificial wetlands, substitutes	broad	OLS, regulating services, Ag landscape focus

CHAPTER 3

VALUING NATIONAL WILDLIFE REFUGES WITH EXISTING META-ANALYSES

In order to explore the usefulness and accuracy of benefit transfers of wetland ecosystem service values using published meta-analysis models, I conduct a Monte Carlo simulation of the distribution of the dependent variable from three existing models: Woodward and Wui (2001); Brander, Florax, and Vermaat (2006); and Ghermandi et al. (2010). The Monte Carlo simulation of each dependent variable's distribution provides a quantification of resampling variability, allowing for the estimation of both mean and median values associated with each meta-analysis model. Because three similar meta-analysis models pertaining to wetland ecosystem services are available, the simulation results allow for the use of a forecast combination approach in order to efficiently combine data from each of the existing meta-analysis studies.

In this chapter, I first discuss the wetland ecosystems in the four case study NWRs. Next, the data and method for estimating and combining the three meta-analysis models are described. The methods I discuss include the Monte Carlo simulation and the forecast combination procedure. Finally, the results of the benefit transfers and forecast combination are presented and discussed.

Description of Four Case Study Refuges

Arrowwood NWR

Figure 2 is a National Land Cover Database (NLCD) 2006 (Fry et al. 2008) map of Arrowwood NWR and surrounding lands. The agricultural context of the NWR is evidenced by the gridded network of roads, the preponderance of cultivated crops and pasture land. The abundance of substitute wetlands is also visible in this map. The lands within the red boundary (representing administratively approved land acquisition boundaries) are either cross-hatched white, designating private ownership or cross-hatched black, designating USFWS ownership. Figure 3 (based on the U.S. Fish and Wildlife Service National Wetland Inventory data set, NLCD 2006 data[1], and the USFWS Cadastral) shows that 29% of approved lands that have been acquired by the USFWS are wetlands. Figure 3 also shows that nearly all wetlands at Arrowwood NWR are emergent or freshwater-marsh wetlands with the majority split between palustrine and lacustrine systems. Additionally, much of the wetlands in Arrowwood NWR are in the vicinity of a riparian ecosystem and are part of the larger Arrowwood National Wildlife Refuge Complex, which includes Arrowwood Wetland Management District, Chase Lake NWR, Chase Lake Wetland Management District, Chase Lake Prairie Project and Valley City Wetland Management District.

The Prairie Pothole region serves as a primary nesting ground supporting extensive populations of economically valuable migratory waterfowl (Niemuth et al. 2006). Accordingly, the economic value of the underlying ecosystem function "provisioning of nesting habitat", aggregated across the region is likely quite large, though this value is not estimated in this dissertation. The location of Arrowwood NWR in the vicinity of numerous other wetlands suggests a decreased welfare impact due to the abundance of substitute wetlands. However, the riparian context of much of Arrowwood's wetlands is a less common wetland feature in the region than pothole wetlands. The substitutability between riparian and pothole wetlands is expected to be greatest for certain services, such as hunting, wildlife observation, and carbon

[1] The pie charts provided for each site depict the distribution of wetlands according to NLCD and NWI data sets, which match the explanatory variables in the Brander, Florax, and Vermaat (2006) and Ghermandi et al. (2010) MA studies, respectively. The only alteration is that for the NLCD distributions, I combine open water with the fresh marsh category as these are the categories included as explanatory variables in the Brander, Florax, and Vermaat (2006) MA.

storage. I expect flood control/storm protection and waste assimilation services to be relatively more valuable for riparian wetlands due to increased hydrological connectivity with downstream populations.

In addition to the differences between riparian and pothole wetlands in ecosystem structure and function, land use history, microclimate, edaphic variation, and microtopography in the Prairie Pothole Region contribute to spatial variation in ecosystem structure and function (Gleason et al. 2011). The effect of ecosystem variation on economic values within the Prairie Pothole context is considered in the meta-analysis benefit transfer only through variations in the distribution of woody vs. non-woody wetlands, the size of the refuge and latitude.

Downstream from Arrowwood NWR the James River flows into the James River Reservoir which provides recreation and flood control/storm protection benefits to Jamestown, South Dakota. The existence of the riparian wetlands and surrounding managed impoundments is expected to lead to delayed and weakened flooding downstream, allowing for higher reservoir levels which benefit recreation services while maintaining the competing service of reduced likelihood and severity of downstream flooding (Cordell and Bergstrom 1993). Flood control/storm protection values are expected to be relatively high because Lake Arrowwood is situated upstream from the Jamestown Reservoir, which provides local recreation benefits and flood control to downstream populations in a region where flood control has been historically important (e.g., DesHarnais et al. 1994). I expect moderate water-quality provisioning benefits due to a lack of nutrient inputs relative to conventional agriculture, and the many downstream beneficiaries of increased water quality.

Blackwater NWR

Blackwater NWR contains extensive wetlands, relatively evenly distributed across woody, herbaceous, and unvegetated wetland land cover classes and with a gradient from freshwater to brackish water. Blackwater NWR is located on the eastern side of the Chesapeake Bay. Figure 4 is an NLCD 2006 map of Blackwater NWR and the surrounding landscape. The map indicates incomplete land acquisitions, with the eastern section of the refuge on the Nanticoke River containing no USFWS acquired land. From the map one can see both the coastal location of Blackwater NWR and the proximity of Cambridge, Maryland north of the western section of the refuge. An abundance of wetlands and agricultural activities can also be seen in the region. Figure 5 demonstrates the abundance of wetlands among the acquired lands of Blackwater NWR, which are split between forested wetlands and brackish marsh. In this figure one can also see that the wetland ecosystem is primarily divided into estuarine and palustrine systems, which is relevant to the Ghermandi et al. 2010 meta-analysis.

Significant research has focused on the Chesapeake Bay and Blackwater NWR, where environmental degradation has been acute and visible as a result of sea-level rise, invasive non-native species, and land-use changes by large populations (Boesch 2007; Kahn and Kemp 1985; Kemp et al. 2005). Management activities at Blackwater NWR occur at a relatively intensive level, including the management of impoundments, and agricultural plots as well as controlled burns. Blackwater NWR is not dominated by lands with wilderness designation, but rather lands that refuge scientists manage for different species. Much of the management is intended to support migratory bird populations, because substitute sites for these populations are decreasing in availability. Key management activities of the Blackwater NWR landscape in support of avifaunal populations include prescribed burns, management of artificial water impoundments, and marsh restoration. Additional management efforts focus on elimination of the introduced, invasive nutria (an aquatic mammal) and restoration of extensive marsh loss partially attributed to the nutria's excessive herbivory. Other management activities include forest plantings, which support forest interior dwelling birds and the endangered Delmarva fox squirrel.

Marsh restoration and construction constitutes an important management input. The goal of marsh restoration and construction is to reverse the loss of an estimated 5,000 acres of marsh since the early twentieth century, according to refuge staff. Another facet of marsh maintenance is the management of invasive species. The invasive nutria as well as mute swans damage existing marsh vegetation such that root mats degrade and soil is removed by water currents. Additionally, the invasive reed *Phragmites* *australis* is also an object of managed eradication efforts, yet refuge biologists acknowledge that invasive marsh species are preferred to open water, a likely alternative if established invasive species are aggressively culled.

Modified landscape features such as Barren Island serve as barriers to storm surge and provide aquatic habitat, and are an important feature of the modern Blackwater ecosystem. Dredge material obtained from the U.S. Army Corps of Engineers, for example, is delivered at no cost to Barren Island where it becomes part of the refuge. Dredged and shipped inputs to the refuge are anthropogenic and contribute toward the economic value of sea level rise protection.

The Chesapeake Bay is in close proximity to large and relatively high-income populations, thus many services are likely to be relatively valuable. I expect that water quality benefits from the Refuge are quite high due to the proximity of large populations and the significant amounts of agricultural inputs upstream from the refuge. Finally, flood control and storm protection benefits are likely high, as Blackwater NWR acts as a barrier to storm surges that might otherwise damage valuable inland properties, such as those in the Cambridge, MD area.

Blackwater NWR is a particularly dynamic site, facing relatively rapid sea-level rise, contributing to the loss of marsh throughout the Chesapeake (Boesch 2007; Kearney, Grace, and Stevenson 1988). Marsh restoration efforts are costly and the durability of restored marshes in an ebb-tide dominated system is questionable (Stevenson et al. 2002). Depending on freshwater and sediment inputs, tidal fluxes, herbivory, subsidence, and prevailing winds, marsh accretion may keep pace with sea-level rise, though marsh loss has been the aggregate long term pattern at Blackwater NWR (Stevenson, Kearney, and Pendleton 1985). Nanticoke estuarine marshes, many which are in the private inholdings classification in Blackwater NWR, have varying accretion rates, with upstream marshes experiencing accretion that exceeds sea-level rise (Ward, Kearney, and Stevenson 1998). Generally, while recent studies of marsh accretion have surprised refuge biologists with the rapidity of accretion and contributed to the evidence of the benefits of prescribed fire to vertical accretion of organic materials (Cahoon et al.

2010), the effects of deep subsidence of land in the area due to post-glacial isostatic rebound compounded with future sea-level-rise “and changes in other climate and environmental drivers (Cahoon et al. 2009)” are indicative of future losses of current marsh lands. Future analysis of ecosystem services in the Blackwater NWR could focus on inclusion of cost-benefit analysis of marsh restoration efforts. The quantitative results generally assume no further loss or gain of wetlands, which is an important assumption in the context of the scientific debate over the magnitude of future sea-level rise.

Okefenokee NWR

The Okefenokee National Wildlife Refuge occupies approximately 400,000 acres, mostly in Southeast Georgia with a small area in Florida. As can be seen in Figure 6, the Okefenokee is dominated by relatively contiguous woody wetlands, and surrounded by extensive patches of discontinuous woody wetlands. The Okefenokee landscape is fed by limited water from uplands resulting in an ombrotrophic or rainfed ecosystem, characterized by scarce nutrients, moderately high salt concentrations, and acidic water (Flebbe 1982). As depicted in Figure 7, approximately 94% of the four hundred thousand acres acquired by the US Fish and Wildlife Service are wetlands. In Figure 7 one can see that the Okefenokee is dominated by woody wetlands in the palustrine system. Wetlands of the Okefenokee have been characterized as closed nutrient systems (Hopkinson 1992) with selective pressure favoring nutrient efficient species.

The Okefenokee is immediately surrounded by a rural landscape with low population densities and relatively low incomes (US Census Bureau 2008). The small town of Waycross, Georgia, population 14,649 (US Census Bureau 2010) is situated to the north of the Okefenokee NWR and Jacksonville, Florida, population 821,784 (US Census Bureau 2010) is approximately 50 km southeast of the refuge. Additionally, according to Refuge staff, people frequently visit the Okefenokee from a variety of distant locations including much of the U.S. as well as Europe.

I expect moderately valuable water-quality provisioning services, as downstream populations are moderately dense and low nutrient water from the Okefenokee tends to dilute nutrient loads from agricultural sources (Katz et al. 1999). Flood control/storm protection benefits are expected to be moderate, as seasonal rains, which might otherwise contribute to downstream flooding, are partially impounded by the Okefenokee Swamp. Additionally, the downstream area of southeast Georgia and Northeast FL experiences frequent and damaging floods due to large rainfall events (Hazards and Vulnerability Research Institute 2012).

Sevilleta and Bosque del Apache NWRs

The Bosque del Apache includes approximately 57,000 acres of USFWS acquired land, matching officially approved acquisition boundaries. The Sevilleta is significantly larger, including approximately 228,000 acres of acquired lands, also with no private inholdings (USFWS 2012). However, as can be seen in Figure 8, due to greater public ownership of Rio Grande river corridor in the Bosque and also due to managed impoundments, the Bosque contains substantially more wetlands. Few substitute wetlands exist in the area and the landscape is dominated by barren land and shrub/scrub. Based on GIS analysis of FWS boundaries and NWI data, the Bosque del Apache NWR and Sevilleta NWR contain an estimated combined 4,958 acres of wetlands, with the Bosque containing the bulk of these wetlands. Throughout this study, wetland valuation results are estimated and reported as an aggregated value across the two refuges. As can be seen in Figure 9, the two refuge system contains only 2% wetlands by surface area, with emergent wetlands constituting the bulk of wetland area. Scrub-shrub land cover dominates the woody wetlands, with only 1% of wetlands identified by NWI data as forested wetlands. About a third of the wetlands are classified as riparian with a small amount of lacustrine wetlands and the remainder in the palustrine system. As the Sevilleta and Bosque del Apache are arid ecosystems, I expect the value of ecosystem services supported by the extensive uplands to be significant: these upland values are not considered in this study.

The study areas in both refuges are along the North American Central Flyway, serving as an important link along the paths of migratory birds where there are few substitute wetlands. In addition to riparian wetlands, The Bosque del Apache NWR contains managed impoundments, which in addition to being managed for water content are cropped via partnerships with local farmers during the spring and summer primarily with corn and alfalfa. Corn and to a lesser extent, alfalfa, serve both to draw migratory waterfowl from surrounding agricultural lands and also as a supplemental source of nutrients for migratory waterfowl.

This meta-analysis does not include benefits of biodiversity or recreation supported by the extensive periodic waterfowl populations in refuge wetlands; further primary valuation or meta-analysis studies are needed to estimate these economic values. Qualitatively, the value of the average wetland is expected to be reduced by low population densities and low state GDP per capita. However, with few wetlands in the region, the lack of substitutes is expected in general to increase the value of refuge wetlands.

I expect that the value of water-quality provisioning services supported by the ecosystem function, nutrient cycling to be moderately high due to the upstream location of significant populations and the pulsed nutrient inputs from agriculture and migratory waterfowl. Kitchell et al. (1999) documents waterfowl nutrient loads and the nutrient sequestration efficiency of Bosque wetlands. Finally, I expect flood control/storm protection benefits to be relatively low due to small downstream populations and the near total control by humans over flooding of the Rio Grande River.

Data

Existing Wetland Ecosystem Service Meta-Analysis Studies

A list of explanatory variables used in the three wetland meta-analysis models can be found in Table 2. Variables are arranged to show similarities across studies, and missing values indicates that variable was not included in the published meta-analysis. For example, Woodward and Wui (2001) do not include variables distinguishing wetland type so this information cannot be included in the meta-analysis benefit transfer associated with that study. All three studies included dummy variables to account for the ecosystem service type being valued ("Services Valued" in Table 2). One can see that while the included variable categories are reasonably consistent across studies, the actual variables used vary considerably. In general, more geographic and socio-economic indicator variables are included in the more recent studies, while method-related and study quality variables decline somewhat in frequency.

I generally anticipate that the accuracy of the set of models increases over time due to methodological and method-related advances and due to the availability of new observations. In the meta-analysis models considered, the list of variables used for describing the socioeconomic and geographic context of each observation generally expands over time, indicating potentially increased precision and a reduced likelihood of biasedness. The inclusion of geospatial variables in the Ghermandi et al. (2010) study describing the population and landscape surrounding each NWR site are examples of advancements expected to increase the accuracy of the meta-analysis model for benefit transfer. Study quality variables are omitted from the two later meta-analysis studies, which Ghermandi et al. (2010) indicate is a result of pre-testing .

Estimated parameter means in each meta-analysis generally match theoretical expectations to the extent that theory offers guidance. The estimated parameters for wetland surface area, for example, indicate a positive impact of adding additional acreage for the average wetland but at a rate that decreases with increasing wetland size. While theory does not specifically predict this result across all wetland sizes and for all services, it is not contrary to my expectations; a large and negative estimated coefficient mean for wetland acreage would be cause for concern. The estimated variances for many parameters can be examined at one time by considering the simulated distribution of WTP, which is discussed in the results below.

As none of the published meta-analysis studies provides a list of observations, interested benefit transfer practitioners are generally left to examine the list of references found at the end of each study. A consequence of this omission is uncertainty about which empirical studies contained in the references were used in modeling and which references appear only because they were cited in the text. The lone exception is the Woodward and Wui (2001) meta-analysis, which contains an internet URL where extensive supplementary information can be obtained, however this information does not appear to precisely match the published model. Knowledge of the overlap in meta-analysis samples would be useful for understanding how predicted benefit transfer results might be correlated across different meta-analysis studies.

Specifying Explanatory Variables for Prediction

To conduct the meta-analysis benefit transfer dependent variable simulation explanatory variables for each transfer must be specified. Next, information from the estimated models is used to program a Monte Carlo simulation that can then be used for assessing benefit transfer accuracy. In addition to using summary information about each meta-analysis model's estimated parameters and variances, an assumption that estimated parameter covariances are zero is required.

The National Wetlands Inventory geospatial database provides the primary definition of wetlands while the NLCD 2006 data set provides landscape data for the wetland type distribution required by the Brander, Florax, and Vermaat (2006) meta-analysis. Refuge boundaries are taken from the USFWS Cadastral Special Interest layers, specifically those parcels designated as "acquired" under the variable, "status" are used.

Population density data are obtained from the SEDAC *Gridded Population of the World: Future Estimates* data set for the year 2010 (CIESIN 2005). GDP data for the state(s) occupied by the NWR are adjusted for inflation to the appropriate data year for each study using the BLS CPI Inflation calculator; these data are from the BEA's *Gross Domestic Product by State* data set (BEA 2010).

While many explanatory variables for meta-analysis benefit transfer can be specified with little difficulty, geospatial and valuation method related variables require important assumptions. Because all three meta-analysis studies estimate non-constant returns to scale, the geographic extent chosen for the analysis will impact the results. As this dissertation focuses on modeling wetlands in NWRs, I use wetlands within NWR boundaries as the unit of analysis. Future work is needed to develop a formal procedure for endogenously identifying the appropriate unit of analysis for wetland ecosystem service modeling, especially in the context of non-constant returns to scale for wetland acreage.

Economic theory provides only limited guidance with respect to the expected impact of primary valuation method on estimated WTP. Best practices for meta-analysis valuation studies often suggest the inclusion of variables controlling for the primary valuation method (e.g., travel cost method, contingent valuation method, etc.) during estimation (Johnston and Rosenberger 2010), while in the course of meta-analysis benefit transfer, primary valuation method-related explanatory variables are often coded at their sample means (Stapler and Johnston 2009), which has the benefit of producing a willingness to pay forecast with the smallest variance. However, because coding method-related explanatory variables as fractions in models with a log-transformed dependent variable, this coding produces a welfare estimate that is a non-linear combination of methods, posing a barrier to criteria validation even if primary valuation studies of all relevant methods were available. Accordingly, I compute meta-analysis benefit transfer estimates coding the method as contingent valuation where applicable in order to include both passive- and active-use benefits in the welfare estimates.

Model B from the Ghermandi et al. (2010) study is used, which omits method-related variables, as the study's authors indicate this is the best model among those estimated. Study quality variables for the Woodward and Wui (2001) meta-analysis were set to zero, implying that the predicted WTP estimates have been obtained from a hypothetical, high-quality primary valuation study.

The appropriate interpretation of the dependent variable is an important and nuanced issue has not been resolved in the ecosystem service meta-analysis benefit transfer literature. In all studies in this field that I am aware of, the dependent variable is treated directly as an estimate of a measure of a WTP construct. This dissertation interprets the dependent variable somewhat differently, as the resulting welfare estimate of a primary valuation study. For forecasts of the dependent variable the value is fundamentally the predicted result of a primary valuation study. Due to the possibility of generating predictions from unlikely combinations of services and methods, such an interpretation better highlights the context of the WTP estimate and implies the appropriate means for criteria validation. For example, one might use a meta-analysis model to predict the value of commercial fishing via the travel cost method at a site where no commercial fishing operations exist; criterion validation via a non-sensical primary valuation study of this predicted value is not feasible. The simulation of the results of such a study is feasible but the validity of the results fails a basic test of content validity.

The data used to estimate flood control benefits from each meta-analysis benefit transfer are given in Tables 3 - 6. The layout of the explanatory variable values matches Table 1. The variables for wetland type must sum to 1, so from Table 2 one can see that Arrowwood NWR is 99.8% fresh marsh (fresh marsh obtained from NLCD 2006 gridcode 95 – emergent herbaceous wetland) and 0.2% woodland (woodland obtained from NLCD 2006 gridcode 90 – forested wetland). The population density variable in the Brander, Florax, and Vermaat (2006) meta-analysis is negative because the units employed are the natural log of 1000people/square kilometer and there are fewer than 1000 people in the average square kilometer in North Dakota. The many dummy variables that were coded to zero are omitted from these tables, but can be seen in Table 2. To predict the value of flood control/storm protection, for example, the flood control/storm protection dummy was set equal to 1 and all other service valued dummy variables were set equal to 0. The analogous coding is used to estimate the value of water quality provisioning.

A general review of the studies referenced in each meta-analysis publication indicates that some observations simultaneously valued more than one service provided by a single wetland site. Specifically, the method-related variables in Brander, Florax, and Vermaat (2006) and Ghermandi et al. (2010) are unlike conventional dummy variables, as they are not perfectly linearly related: one valuation study may use multiple methods to obtain a result. Because no interaction effects among different services are estimated in any of the models, the resulting restriction is that the partial effect of predicting WTP for a single service provided by a wetland (e.g., flood control) is the same as adding that same service to another (e.g., adding flood control to a benefit transfer value that also predicts water quality provisioning value); this restriction can imply negative partial effects for certain services. For example, with the Brander, Florax, and Vermaat (2006) meta-analysis, valuing flood control/storm protection alone will always return a positive WTP estimate, but simultaneously valuing flood control/storm protection and water quality will have a lower predicted value, indicating that water quality services have a negative effect. I handle this complication by always valuing only one service at a time.

Method

Once data is in hand, the first step in the method is a parametric Monte Carlo simulation of the dependent variables of interest for each meta-analysis. The second step is a forecast combination procedure that combines the results from the first step for each service at each refuge.

Monte Carlo Simulation of Distribution

As mentioned above, the popularity of using logged dependent variables necessitates a more complex approach such as bootstrapping or Monte Carlo simulation to capture the effects of curvature of the exponential function within the expected value operator (Wooldridge 2002). A bootstrapping approach would be desirable, but is not possible without the original sample.

The original meta-analysis studies assessed in this chapter do not report the full covariance matrix of the estimated parameters, so I assume zero covariance between parameters, resulting in a diagonal variance-covariance matrix. The basic algorithm for the parametric Monte Carlo simulation, implemented in Matlab, is to draw a pseudo random vector of parameter values associated with each meta-analysis using a multivariate normal distribution with mean and variance equal to the reported parameters. Each of the constant value regressors found in Tables 3 - 6 is multiplied by the appropriate randomly drawn vector of parameters and the sum of these products is exponentiated by the base of the natural logarithm. The process is iterated one million times for each meta-analysis, each wetland NWR and each of the two services considered, resulting in 24 vectors with length equal to one million containing log normally distributed WTP estimates. I chose to use a large number of iterations after pretesting with fewer iterations led to unstable estimates of the mean and variance. The mean of each series and the variance of the mean are computed, and from each of the 24 sorted vectors I obtain quantiles representing the 5th, 50th, and 95th percentiles.

The parametric Monte Carlo simulation is useful because it provides a means for estimating the variance of each WTP estimate. The popular alternative procedure typically requires an estimate of the standard error of the residual (Wooldridge 2002), which was not available for all the meta-analysis studies considered. The assumption that the covariances of the parameters are all zero may lead to substantial error in the estimated variance of the dependent variable; in particular, this may be a problem when used with a model based on a small sample. For example, if two variables have positive values and positive estimated coefficients with negative covariance and this covariance is assumed to be zero, the resulting sum of these variables will be biased upwards. As the purpose of the experiment is to demonstrate how to implement meta-analysis benefit transfer using a suite of existing models with published results, and I am unaware of a superior use of the available information, I rely on the assumption of non-correlated parameter estimates.

Forecast Combination

An inverse variance weight approach (DerSimonian and Laird 1986; Borenstein et al. 2009) combines data from the simulated distribution of WTP into a single value. The forecast combination literature suggests an inverse variance weighting approach often leads to improved forecast performance

(J. Smith and Wallis 2009). The point estimate of the median offers a measure of central tendency for the forecast combination procedure less sensitive to assumptions required to calculate a mean from the Monte Carlo simulation model. Aside from requiring weaker assumptions, median values are useful as a measure of central tendency because they may better reflect the outcome of a democratic referendum and because the influence of large outliers is reduced. Additionally, as the logarithmic transformation of WTP ensures positive predicted values across the simulated distribution (censoring of non-positive WTP estimates), the use of the median measure of central tendency opposes to some extent the effect of a truncated predicted distribution (i.e., negative values are not allowed when the dependent variable is in log form). Accordingly, the forecast combination employed in this chapter takes a weighted average of the three median values estimated from each meta-analysis for each service at each refuge. Interestingly, the inverse variance weighting forecast combination is identical to implementing an efficient generalized least squares (GLS) regression (Wooldridge 2002) with only an intercept and with known variances. The weighted average of WTP for site i and service j, can be seen in equation (7), where subscripts i and j index the site and service and subscript k indexes the meta-analysis used for the WTP estimate. Summation is over k=1,…,3 for the three meta-analysis studies.

$$\overline{WTP}_{ij} = \frac{\Sigma\left[\frac{1}{V_{ijk}} med(WTP_{ijk})\right]}{\Sigma\left[\frac{1}{V_{ijk}}\right]} \tag{7}$$

In equation (7) the point estimate or median of the distribution serves as a conservative central estimate of WTP and the simulated variance of the sample mean serves as a measure of resampling variability of the estimate of the median. In equation (1), each weight is then the inverse of the variance of the mean associated with that observation and the sum is normalized by the sum of the inverse variances associated with each meta-analysis model. Alternatively, inverse variances can be interpreted as precision estimates and each weight is the share of that observation's precision among the three meta-analysis studies. In addition to the inverse variance or precision weighted average, I also estimate a conventional, evenly weighted average to illustrate the impact of the inverse variance weights.

Results

The results of the Monte Carlo simulation of WTP per acre can be found in Tables 7 to 12. Tables 7 to 10 contain the results of one million multivariate normal parameter draws from the Monte Carlo analysis. Each table contains the valuation results for one of the case study refuges. The results for flood control/storm protection are in the top half of each table and the results for water quality provisioning are in the bottom half. Each row of Table 6 to 9 contains an estimate of WTP per year per hectare or acre from a particular meta-analysis regression model. The last two columns within the rows for each service are the precision and unweighted averages of the three estimates above. One can see that med(WTP) converges to $e^{E[X\beta]}$. The convergence of the median of the simulated distribution and the dependent variable point estimate from each meta-analysis regression equation is a basic feature of the lognormal distribution based on a symmetric normal distribution[2]. Tables 11 and 12, respectively contain the results for flood control/storm protection and water quality. The first numerical column is the estimate of the mean of the dependent variable after being exponentiated by the antilogarithm. The second column of numerical data is the simulated variance of the mean and the rows contain each of the NWRs and then each meta-regression. A comparison of the dispersion of WTP estimates from each meta-analysis, indicated clearly both by both the variance estimates and the 5th and 95th estimated percentiles, suggests that the Ghermandi et al. (2010) meta-analysis is far more precisely estimated than the two older meta-analysis studies. While this conclusion might be a result of estimation error due to the assumption that all sample covariances among parameters were equal to zero, with limited information the Ghermandi et al. (2010) meta-analysis appears to be the most precise by a substantial margin. Accordingly, I suggest that the Ghermandi et al. (2010) meta-analysis values are the best estimates available from the three existing

[2] As bootstrapping procedures do not ensure symmetry, the median of a bootstrapping simulation may not match the point estimate as it does with a parametric simulation of a normally distributed variable.

meta-analysis models. The Ghermandi et al. (2010) benefit estimates can be used in place of the forecast combination results without a loss of accuracy, as the introduction of the median values estimated with the Brander, Florax, and Vermaat (2006) and the Woodward and Wui (2001) meta-analysis studies have a negligible effect on the combined forecasts. This conclusion may be sensitive to the assumptions about the estimated parameter covariances required for the Monte Carlo simulation.

The point estimates in Tables 11 and 12 are most similar between the Brander, Florax, and Vermaat (2006) and Ghermandi et al. (2010) models. The Woodward and Wui (2001) values are substantially larger than the equivalently coded forecasts from the other two studies. This divergence can be mostly attributed to the inclusion in the Woodward and Wui (2001) meta-analysis of study quality variables, which indicate that lower quality studies systematically produce lower welfare estimates. Because the forecasts made with the Woodward and Wui (2001) model summary are based on hypothetical high quality valuation studies, these estimates are higher than otherwise. Figure 10 is a bar graph containing point estimates of WTP per acre per year for each service at each refuge and estimated with each meta-analysis model. In Figure 10, the left hand axis corresponds to the blue bars for the Woodward and Wui (2001) point estimates while the right hand axis corresponds to the green and red bars for the Ghermandi et al. (2010) and Brander, Florax, and Vermaat (2006) studies. An important feature of the welfare estimates concerns the differences across meta-analysis models in the relative WTP estimates for each refuge. While both the Brander, Florax, and Vermaat (2006) meta-analysis and the Ghermandi et al. (2010) meta-analysis place Blackwater NWR as having the most valuable wetlands, The Woodward and Wui (2001) meta-analysis places Blackwater NWR as third most valuable per unit of land. This result is due in large part to the inclusion of socio-economic variables in the later two meta-analysis models.

Next, WTP estimates are aggregated over all wetland acres in the refuge for each service. Figures 11 and 12 are graphic representations of the estimates of WTP for ecosystem services supported by all NWI identified wetlands in each refuge. The corresponding numerical aggregate values can be found in

Table 13 along with median (obtained via point estimate) WTP and the total count of NWI identified wetlands within each refuge. As discussed above the Monte Carlo simulation indicates that the best estimates of WTP are those obtained from the point estimate method of obtaining the median from the Ghermandi et al. (2010) meta-analysis WTP distribution, found in the last 8 rows of Table 13.

The results in Table 13 and Figures 10, 11, and 12 generally indicate that for wetland ecosystem services the Okefenokee NWR is most valuable as a whole with the Blackwater NWR providing moderately less valuable ecosystem services. The large estimated values of the services associated with Okefenokee NWR wetlands are due largely to the refuge's extensive wetland surface area. The aggregate values are large despite the fact that median values per acre estimated for the Okefenokee NWR wetland ecosystem are among the lowest of the four refuges considered across all three meta-analysis studies. The values of the average acre of the Okefenokee NWR are lowest primarily due to the decreasing returns to scale relationship estimated in all three meta-analysis studies. For both Arrowwood NWR and Sevilleta and Bosque del Apache NWRs aggregate values of wetland ecosystem services are estimated to be substantially lower than the other refuges, by a factor of approximately ten. The primary reason for these lower values is the limited surface area of wetlands in these two refuges. The high values per average acre of Blackwater NWR wetlands are due largely to the high incomes and population densities coupled with modest surface area or scale.

Discussion

A considerable degree of uncertainty exists with regard to the accuracy and suitability of meta-analysis benefit transfer estimates as ecosystem service values. The large variances associated with the Monte Carlo simulation are indicative of a large degree of inaccuracy. The existence of heteroskedasticity in the model suggests that certain predicted values might be more precisely estimated than others. For example, Figure 5 in the Brander, Florax, and Vermaat (2006) meta-analysis suggests that the model overvalues high-value wetlands and undervalues low-value wetlands and that moderately high valued wetlands have the lowest percent transfer error while the lowest value wetlands have the highest percent transfer error. Unfortunately limited information about this type of heteroskedasticity prevents one from forming more concrete expectations about the accuracy of predicted valation results. I suggest that the results of meta-analysis benefit transfers using model summaries are most apt for scoping decisions or for evaluating the likelihood that inclusion of ecosystem service values are likely to change the results of a benefit-cost analysis. When meta-analysis benefit transfer forecasted ecosystem service values do suggest that the optimal decision hinges on ecosystem service values, primary valuation studies may be warranted.

Generally, I wish to emphasize the existence of two possible interpretations of the dependent variable – convention is to directly interpret the predicted dependent variable as a measure of WTP. The more direct interpretation of the dependent variable estimate is that the values are a forecast of the result of a hypothetical primary valuation study. This hypothetical study interpretation provides some guidance with respect to coding method-related or “nuisance” (Moeltner, Boyle, and Paterson 2007) variables as well as for performing criteria validation (Bishop 2003). While not explored in this chapter, the use of sample means for the values of binary method-related variables under this interpretation is less appropriate than the weighting of multiple, single-method forecasts (from a single meta-analysis model) by sample means or estimated parameter variances (a within rather than across study forecast combination approach); mathematically, the issue is analogous to the divergence between median and mean WTP for a log-linear regression model. Specifically, by Jenson’s inequality, the within study forecast combination approach will lead to larger welfare estimates, which may be tempered by excluding methods from the combined forecasts or by further modifying the weights for each method to incorporate the variance of that approach, a broader forecast combination procedure than this chapter presents. Lacking sample means for each study, this more statistically robust forecast combination procedure cannot be performed. However, value estimates that combine multiple valuation methods that are applicable to different aspects of economic value create problems for interpreting and applying these values. Because this study forecasts CVM study results, the predicted values are applicable to questions relating to the combined passive and active use values to society.

The heterogeneity of the primary data for each meta-analysis model used in the novel analysis suggests that one should question whether the best interpretation of predicted values is that they are the value of a service or the value of a wetland that provides a service. In choosing the contingent valuation method (CVM), the most apt interpretation is that estimated values apply to a wetland that provides a service. A review of many of the CVM studies that are referenced by each meta-analysis indicates that the valuation question is typically focused on a particular wetland extent and the respondents are notified of the most important services provided by that wetland. The units of the dependent variable, WTP/area/year also suggest that the appropriate interpretation of the meta-analysis model dependent variable is associated with a wetland that provides ecosystem services.

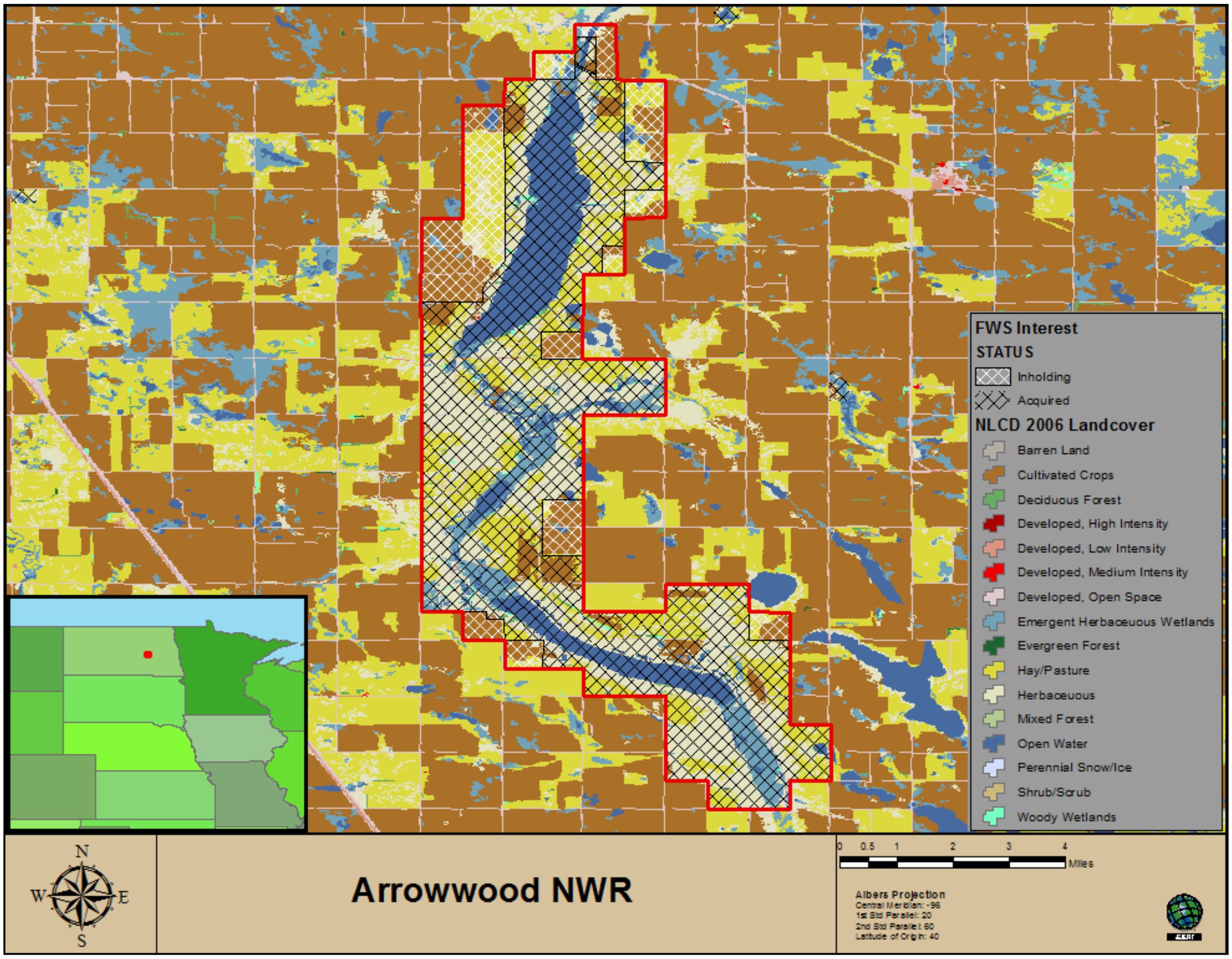


Figure 2: NLCD 2006 map of Arrowwood NWR

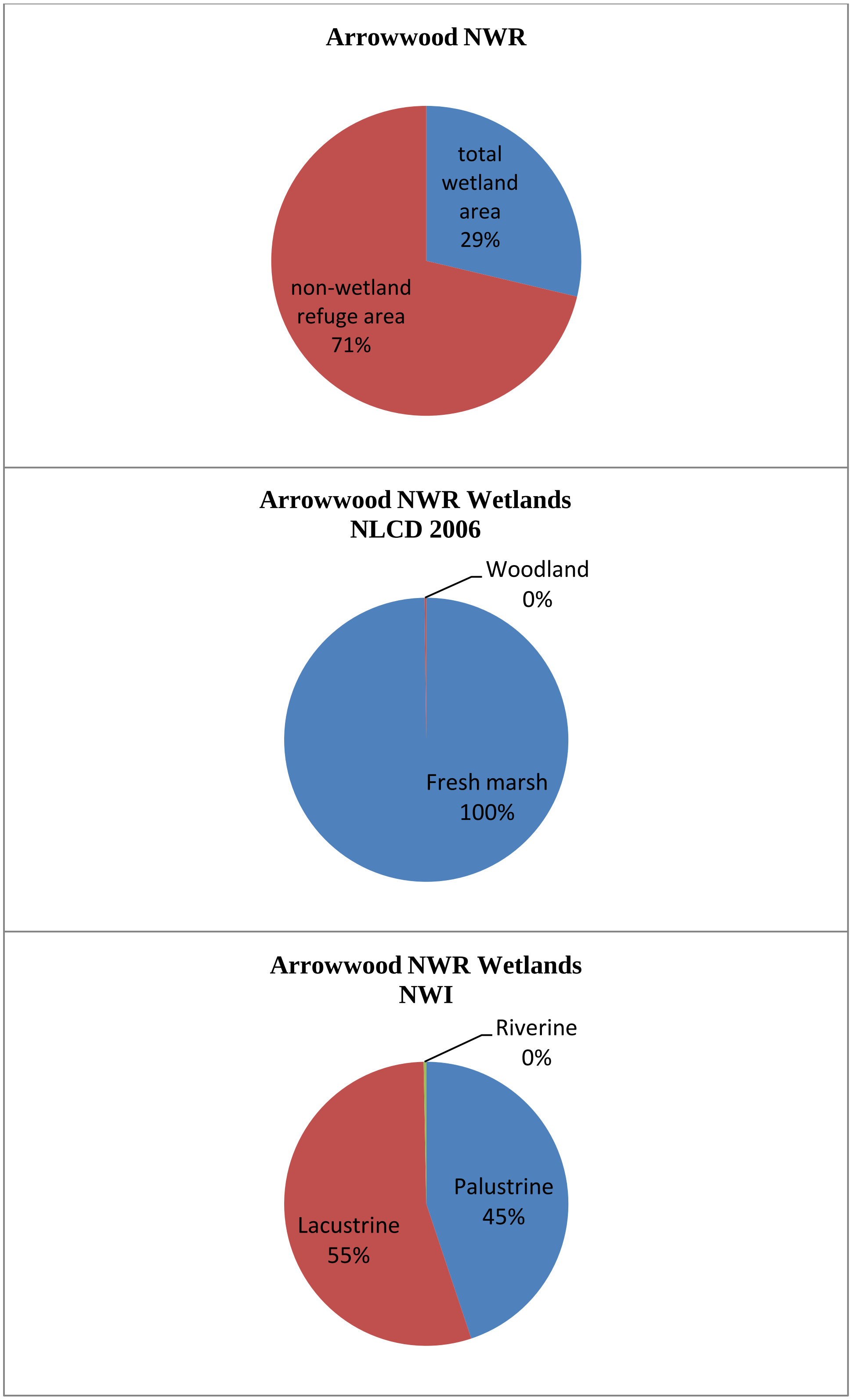


Figure 3: Arrowwood NWR wetland distribution

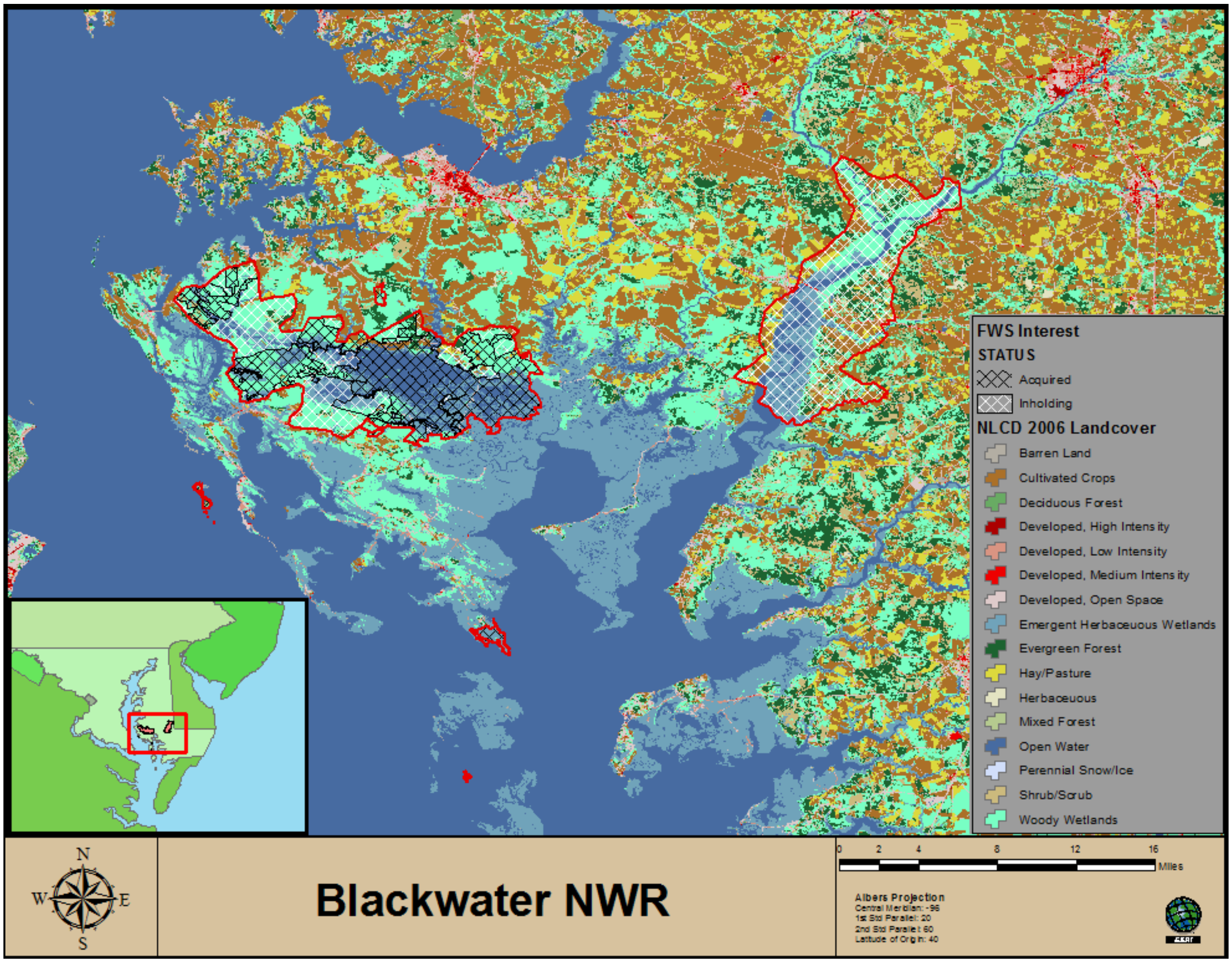


Figure 4: NLCD 2006 map of Blackwater NWR

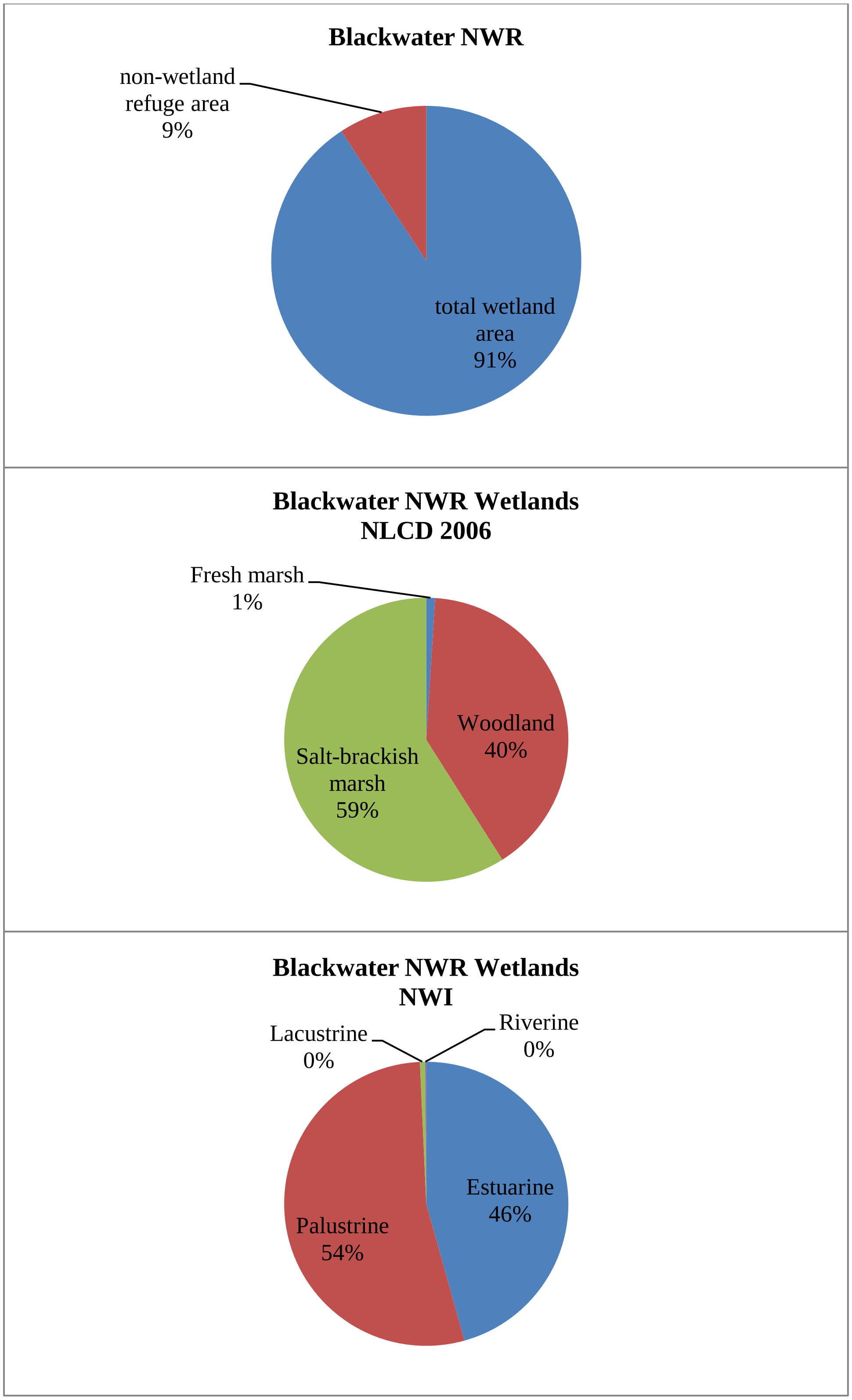


Figure 5: Blackwater NWR wetland distribution

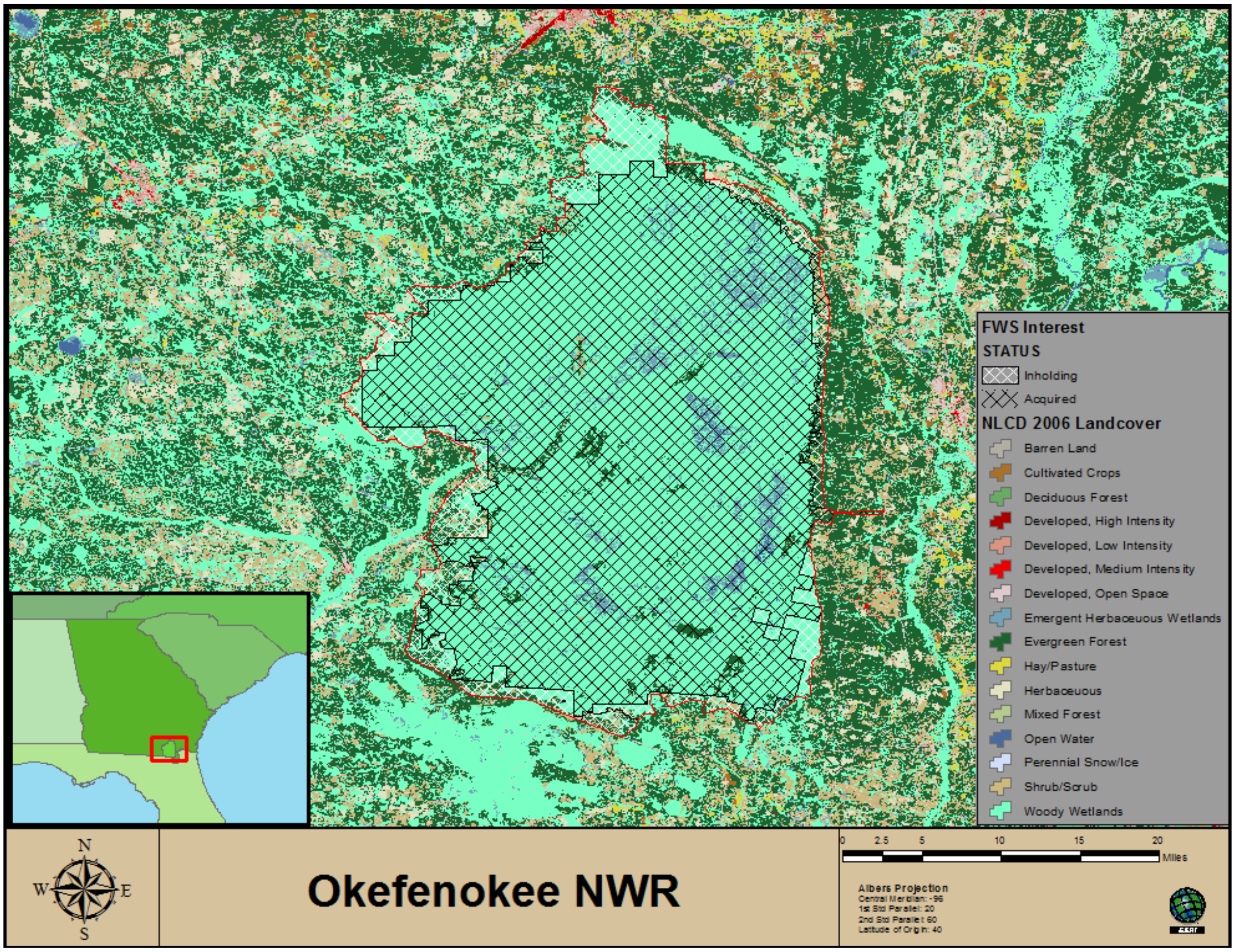


Figure 6: NLCD 2006 map of Okefenokee NWR

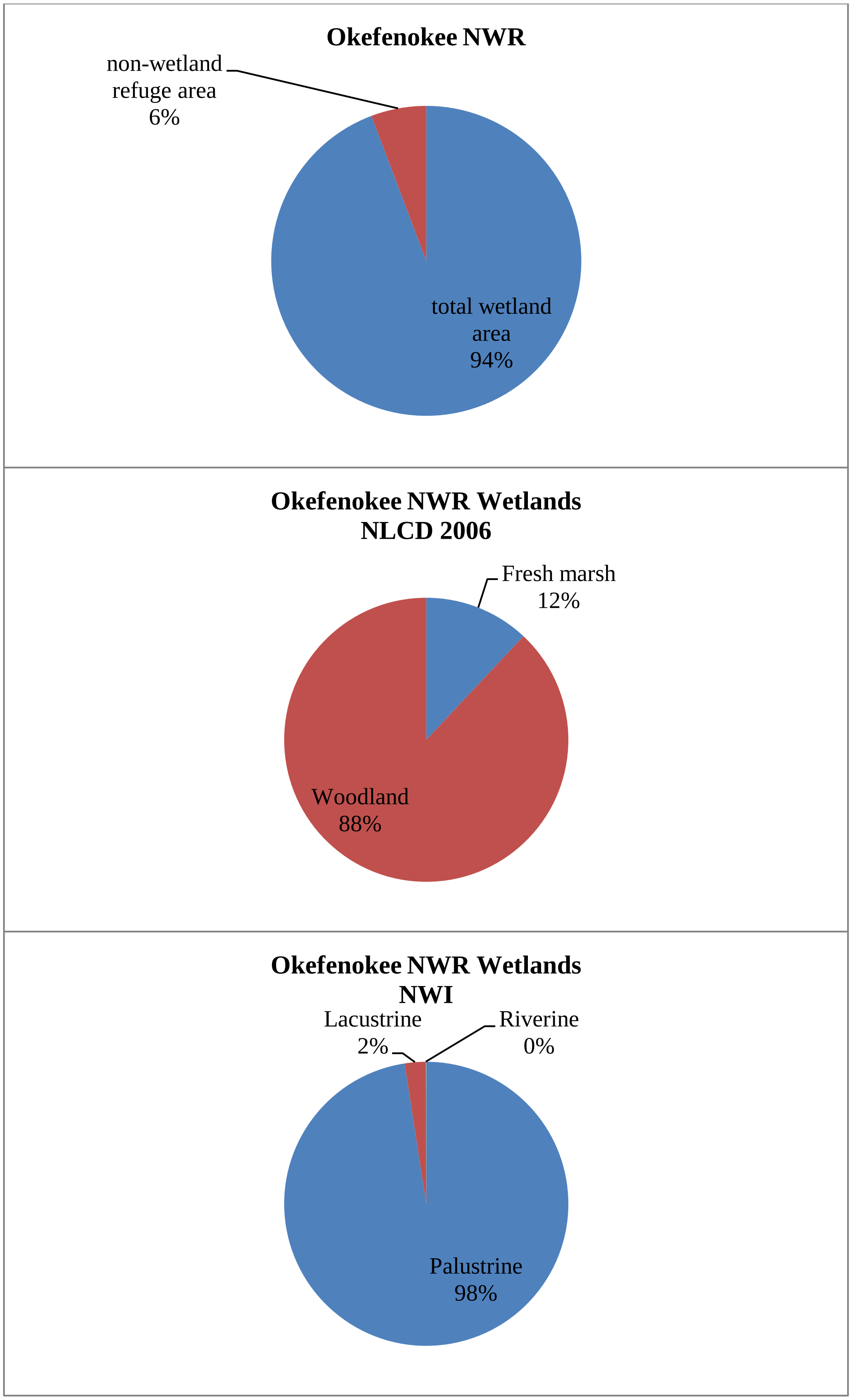


Figure 7: Okefenokee NWR wetland distribution

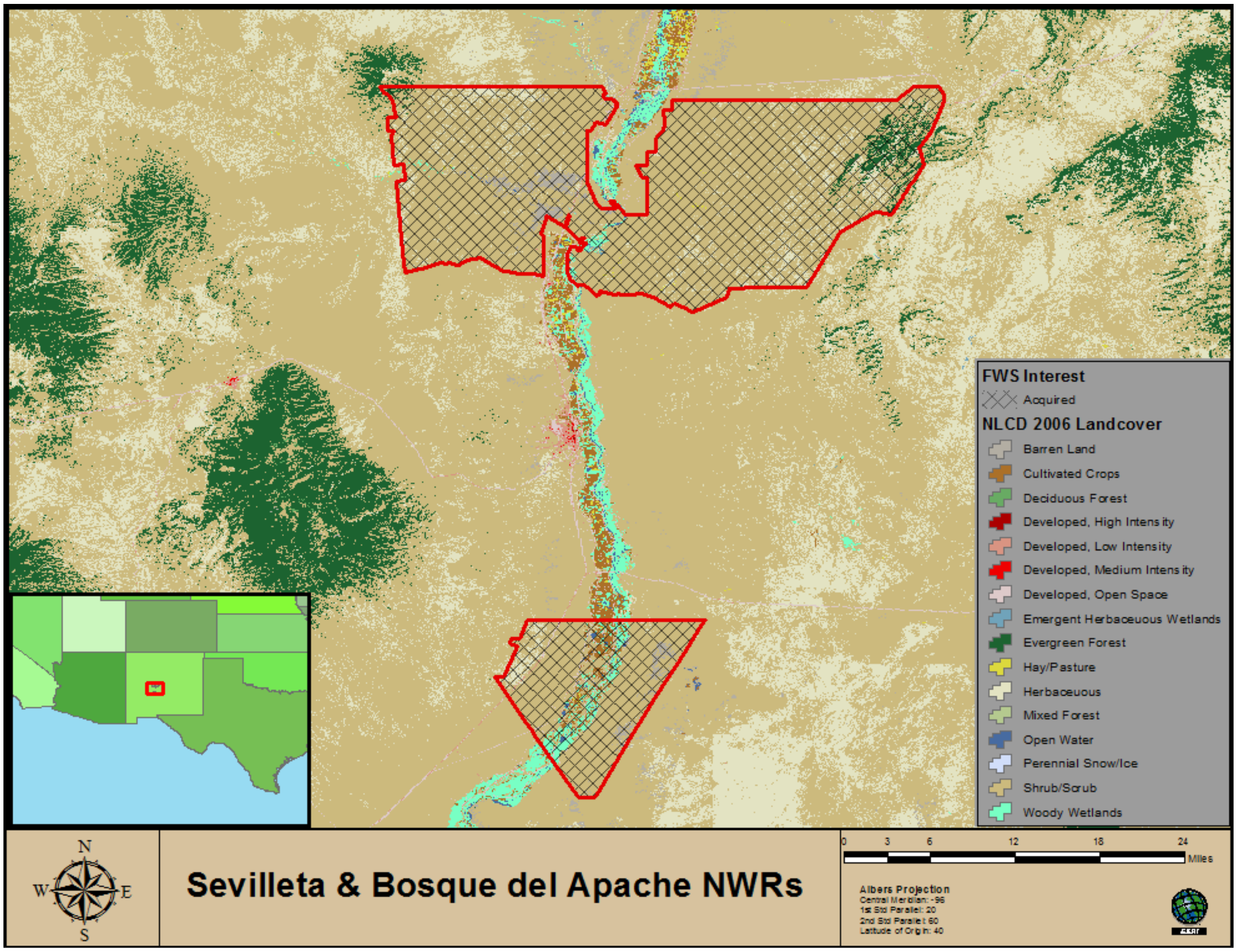


Figure 8: NLCD 2006 map of Sevilleta and Bosque del Apache NWRs

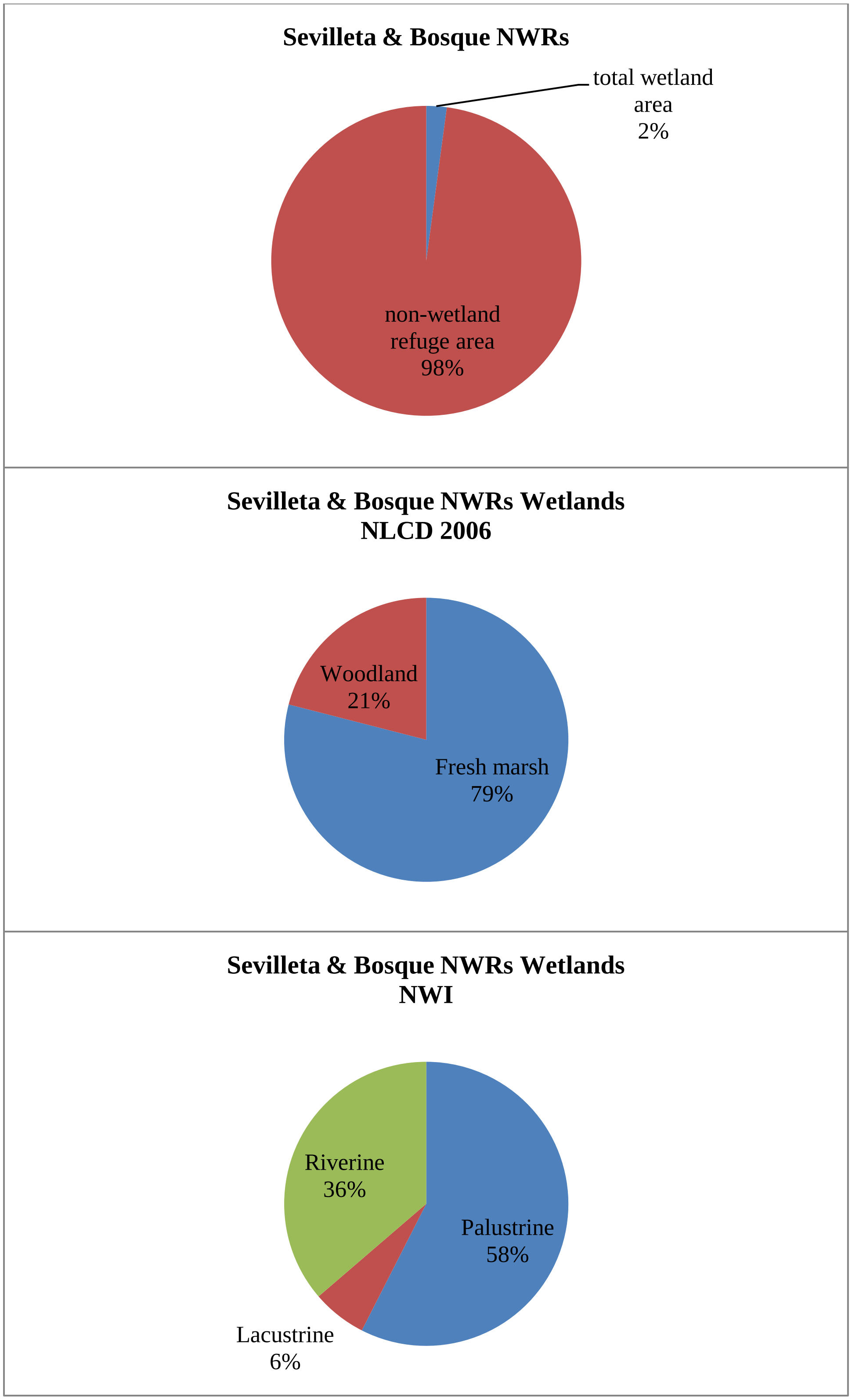
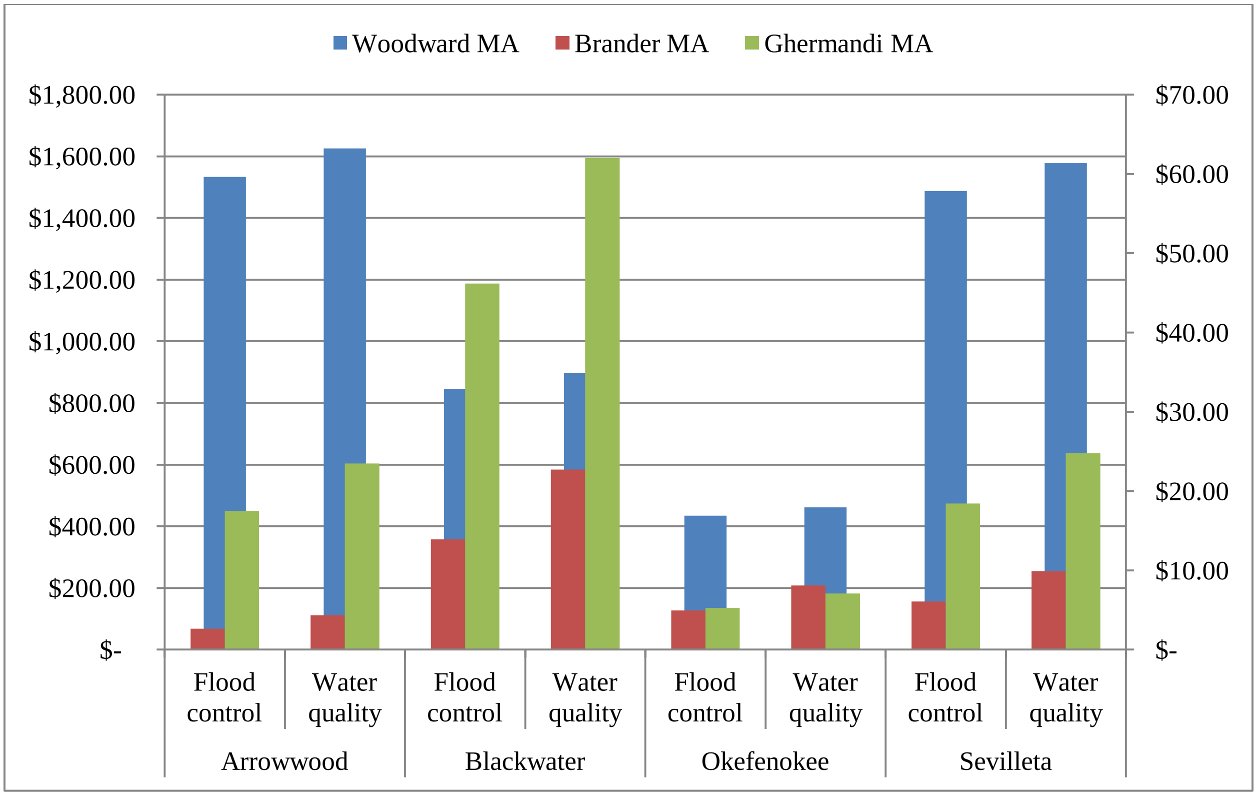
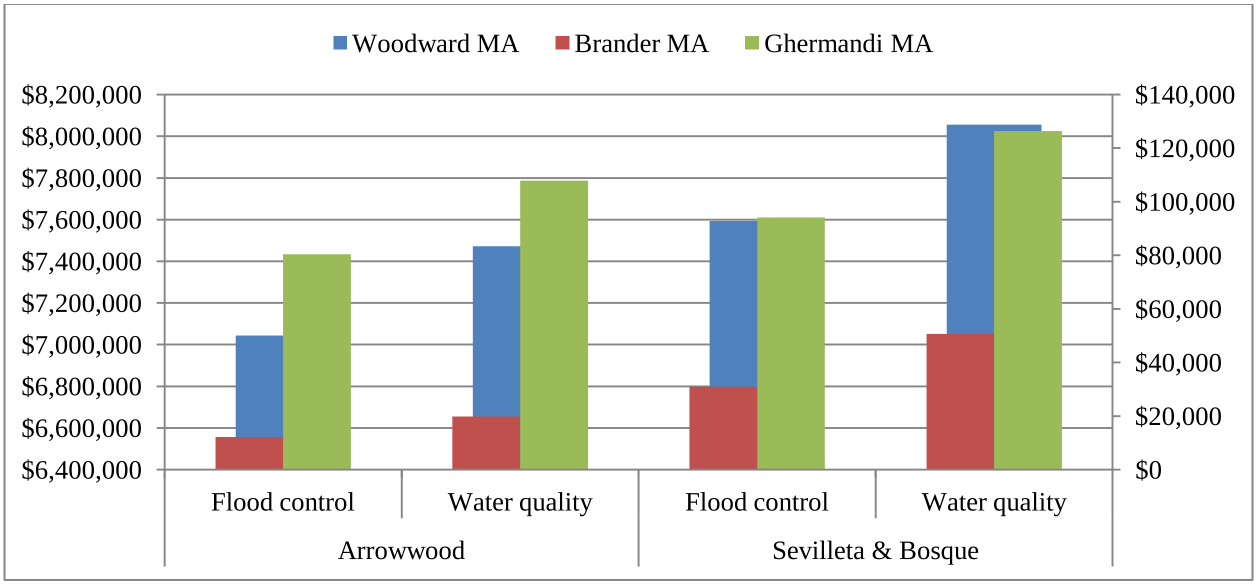


Figure 9: Sevilleta and Bosque NWRs wetland distribution



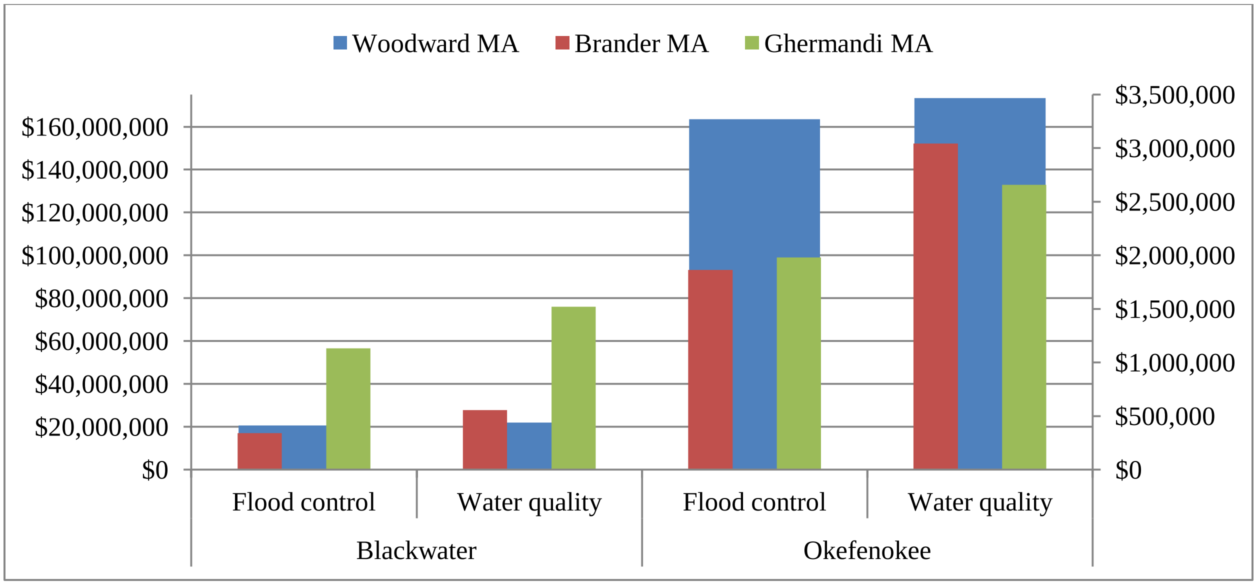
Woodward and Wui 2001 values correspond to left side axis labels and Brander, Florax, and Vermaat (2006) and Ghermandi et al. (2010) values correspond to right side axis labels, all values 2010 US dollars per average acre per year

Figure 10: Point estimates of WTP per average acre by NWR wetlands and service



Woodward and Wui 2001 values correspond to left side axis labels and Brander, Florax, and Vermaat (2006) and Ghermandi et al. (2010) values correspond to right side axis labels, all values 2010 US dollars for all refuge wetlands per year

Figure 11: Arrowwood and Sevilleta and Bosque NWR point estimates of WTP for wetland ecosystem services



Woodward and Wui 2001 values correspond to left side axis labels and Brander, Florax, and Vermaat (2006) and Ghermandi et al. (2010) values correspond to right side axis labels, all values 2010 US dollars for all refuge wetlands per year

Figure 12: Blackwater and Okefenokee NWR point estimates of WTP for wetland ecosystem services

Table 2: Variables used in three published wetland ecosystem services meta-analysis studies

	Woodward and Wui (2001) Model C	Brander, Florax, and Vermaat (2006)	Ghermandi et al. (2010) Model B
Landscape	Log (acres)	Log (hectares) Absolute value(latitude) $Latitude^2$ % of wetland identified as: Mangrove Unvegetated sediment Salt-brackish marsh Fresh marsh Woodland	Log (hectares) % of wetland identified as: Estuarine Marine Riverine Palustrine Lacustrine Human made dummy Wetland area in 50km radius
Geog. and socio-economic	Coastal indicator	Log (GDP per capita) Log (population density) Study location indicators: South America Europe Asia Africa Australasia Urban Ramsar proportion	Medium-low human pressure Medium-high human pressure High human pressure GDP per capita Population in 50km radius
Method-related	Year of publication Valuation method indicators: Hedonic Travel cost Replacement cost Net factor income Producer surplus	Marginal Valuation method indicators: Hedonic Travel cost Replacement cost Net factor income Contingent valuation Production function Market prices Opportunity cost	Marginal Year of publication
Service valued	Service valued indicator: Flood control Groundwater recharge Water quality Commercial fishing Recreational fishing Amenity Erosion reduction Bird watching Non-use appreciation of species	Service valued indicator: Flood control Water supply Water quality Commercial fishing and hunting Recreational fishing Recreational hunting Amenity Fuel wood Biodiversity Material	Service valued indicator: Flood control Surface/groundwater supply Water quality Comm. fishing and hunting Recreational fishing Recreational hunting Amenity and aesthetics Fuel wood Natural habitat, biodiversity Harvesting of natural materials Nonconsumptive recreation
Study quality	Indicator variables noting: Published results Poor data quality Poor theory Poor econometrics		

Table 3: Variable values for Arrowwood NWR Flood Control Meta-Analysis Benefit Transfer with each published meta-analysis

Arrowwood NWR - Flood Control						
	Woodward and Wui (2001) Model C		Brander, Florax, and Vermaat (2006)		Ghermandi et al. (2010) Model B	
	Variable	*Value*	*Variable*	*Value*	*Variable*	*Value*
	Intercept	= 1	Intercept	= 1	Intercept	= 1
Landscape	Log acres	= 8.43	Log hectares	= 7.53	Log hectares	= 7.53
	-	-	Abs. val. Lat.	= 47.24	-	-
	-	-	Latitude^2	= 2231	-	-
Wetland type (ratios)	-	-	Fresh marsh Woodland All others	= 0.998 = 0.002 = 0	Palustrine Lacustrine Riverine All others	= 0.45 = 0.55 = 0.003 = 0
Socio-economic	-	-	Log (GDP per capita)	= 10.41	Log (GDP per capita)	= 10.59
	-	-	Log (Pop. Den.)	= -5.89	Log(Pop. in 50km)	= 9.99
	-	-	Ramsar Proportion	= 0	Human pressure indicators	= 0
Geog. indicators	Coastal	= 0	All	= 0	-	-
Method-ological	-	-	Marginal	= 0	Marginal	= 0
	Year of publication	= 52	-	-	Year of publication	= 36
Valuation method indicators	All	= 0	CV All others	= 1 =0	-	-
Services valued indicators	Flood control All others	= 1 = 0	Flood control All others	= 1 = 0	Flood control All others	= 1 = 0
Study quality indicators	All	= 0	-	-	-	-

Table 4: Variable values for Blackwater NWR Flood Control Meta-Analysis Benefit Transfer with each published meta-analysis

Blackwater NWR - Flood Control						
	Woodward and Wui (2001) Model C		Brander, Florax, and Vermaat (2006)		Ghermandi et al. (2010) Model B	
	Variable	*Value*	*Variable*	*Value*	*Variable*	*Value*
	Intercept	= 1	Intercept	= 1	Intercept	= 1
Landscape	Log acres	= 10.11	Log hectares	= 9.20	Log hectares	= 9.20
	-	-	Abs. val. Lat.	= 38.4	-	-
	-	-	Latitude^2	= 1474.6	-	-
Wetland type (ratios)	-	-	Fresh marsh Woodland Salt-brackish marsh All others	= 0.01 = 0.40 = 0.59 = 0	Estuarine Palustrine Lacustrine Riverine	= 0.23 = 0.27 = 0.003 = .0006
Socio-economic	-	-	Log (GDP per capita)	= 10.55	Log (GDP per capita)	= 10.73
	-	-	Log (Pop. Den.)	= -3.47	Log(Pop. in 50km)	= 12.65
	-	-	Ramsar Proportion	= 1	Human pressure indicators	= 0
Geog. indicators	Coastal	= 0	All	= 0	-	-
Method-ological	-	-	Marginal	= 0	Marginal	= 0
	Year of publication	= 52	-	-	Year of publication	= 36
Valuation method indicators	All	= 0	CV All others	= 1 =0	-	-
Services valued indicators	Flood control All others	= 1 = 0	Flood control All others	= 1 = 0	Flood control All others	= 1 = 0
Study quality indicators	All	= 0	-	-	-	-

Table 5: Variable values for Okefenokee NWR Flood Control Meta-Analysis Benefit Transfer with each published meta-analysis

Okefenokee NWR - Flood Control						
	Woodward and Wui (2001) Model C		Brander, Florax, and Vermaat (2006)		Ghermandi et al. (2010) Model B	
	Variable	*Value*	*Variable*	*Value*	*Variable*	*Value*
	Intercept	= 1	Intercept	= 1	Intercept	= 1
Landscape	Log acres	= 12.84	Log hectares	= 11.9	Log hectares	= 11.9
	-	-	Abs. val. Lat.	= 30.84	-	-
	-	-	Latitude^2	= 950.92	-	-
Wetland type (ratios)	-	-	Fresh marsh Woodland All others	= 0.12 = 0.88 = 0	Palustrine Lacustrine Riverine All others	= 0.976 = 0.024 = .0004 = 0
Socio-economic	-	-	Log (GDP per capita)	= 10.27	Log (GDP per capita)	= 10.45
	-	-	Log (Pop. Den.)	= -5.60	Log(Pop. in 50km)	= 10.95
	-	-	Ramsar Proportion	= 1	Human pressure indicators	= 0
Geog. indicators	Coastal	= 0	All	= 0	-	-
Method-ological	-	-	Marginal	= 0	Marginal	= 0
	Year of publication	= 52	-	-	Year of publication	= 36
Valuation method indicators	All	= 0	CV All others	= 1 =0	-	-
Services valued indicators	Flood control All others	= 1 = 0	Flood control All others	= 1 = 0	Flood control All others	= 1 = 0
Study quality indicators	All	= 0	-	-	-	-

Table 6: Variable values for Sevilleta and Bosque del Apache NWRs Flood Control Meta-Analysis Benefit Transfer with each published meta-analysis.

Sevilleta and Bosque del Apache NWRs - Flood Control						
	Woodward and Wui (2001) Model C		Brander, Florax, and Vermaat (2006)		Ghermandi et al. (2010) Model B	
	Variable	*Value*	*Variable*	*Value*	*Variable*	*Value*
	Intercept	= 1	Intercept	= 1	Intercept	= 1
Landscape	Log acres	= 8.54	Log hectares	= 7.63	Log hectares	= 7.63
	-	-	Abs. val. Lat.	= 34.05	-	-
	-	-	Latitude^2	= 1159.6	-	-
Wetland type (ratios)	-	-	Fresh marsh Woodland All others	= 0.79 = 0.21 = 0	Palustrine Lacustrine Riverine All others	= 0.575 = 0.062 = 0.363 = 0
Socio-economic	-	-	Log (GDP per capita)	= 10.24	Log (GDP per capita)	= 10.42
	-	-	Log (Pop. Den.)	= -5.45	Log(Pop. in 50km)	= 10.24
	-	-	Ramsar Proportion	= 0	Human pressure indicators	= 0
Geog. indicators	Coastal	= 0	All	= 0	-	-
Method-ological	-	-	Marginal	= 0	Marginal	= 0
	Year of publication	= 52	-	-	Year of publication	= 36
Valuation method indicators	All	= 0	CV All others	= 1 =0	-	-
Services valued indicators	Flood control All others	= 1 = 0	Flood control All others	= 1 = 0	Flood control All others	= 1 = 0
Study quality indicators	All	= 0	-	-	-	-

Table 7: Arrowwood NWR Meta-Analysis Benefit Transfer Monte Carlo quantiles

	Arrowwood NWR	Predicted per-acre value, $e^{E[X\beta]}$	Simulated distribution of expected per-acre value		
			5th percentile	50th percentile	95th percentile
Flood Control	Woodward and Wui	1532.83	4.56	1528.94	513829.36
	Brander, Florax, and Vermaat	2.65	0.0000005	2.61	14011026.33
	Ghermandi et al.	17.48	0.19	17.56	1634.44
	Precision weighted average	17.49			
	Unweighted average	517.66			
Water Quality	Woodward and Wui	1625.99	4.89	1621.53	544059.38
	Brander, Florax, and Vermaat	4.33	0.0000008	4.28	23459222.04
	Ghermandi et al.	23.48	0.25	23.58	2176.20
	Precision weighted average	23.49			
	Unweighted average	551.27			

Table 8: Blackwater NWR Meta-Analysis Benefit Transfer Monte Carlo quantiles

	Blackwater NWR	Predicted per-acre value, $e^{E[X\beta]}$	Simulated distribution of expected per-acre value		
			5th percentile	50th percentile	95th percentile
Flood Control	Woodward and Wui	844.86	1.90	843.77	370704.19
	Brander, Florax, and Vermaat	13.91	0.0000057	13.75	34432273.25
	Ghermandi et al.	46.16	0.42	46.43	5155.70
	Precision weighted average	46.17			
	Unweighted average	301.64			
Water Quality	Woodward and Wui	896.22	2.05	893.67	392006.84
	Brander, Florax, and Vermaat	22.70	0.0000090	22.38	58489215.98
	Ghermandi et al.	62.00	0.56	62.36	6918.91
	Precision weighted average	62.03			
	Unweighted average	326.97			

Table 9: Okefenokee NWR Meta-Analysis Benefit Transfer Monte Carlo quantiles

	Okefenokee NWR	Predicted per-acre value, $e^{E[X\beta]}$	Simulated distribution of expected per-acre value		
			5th percentile	50th percentile	95th percentile
Flood Control	Woodward and Wui	435.00	0.89	433.94	209566.46
	Brander, Florax, and Vermaat	4.96	0.0000037	4.91	6702403.62
	Ghermandi et al.	5.27	0.05	5.29	550.86
	Precision weighted average	5.27			
	Unweighted average	148.41			
Water Quality	Woodward and Wui	461.42	0.97	460.82	221840.58
	Brander, Florax, and Vermaat	8.09	0.0000058	8.01	11151922.26
	Ghermandi et al.	7.07	0.07	7.11	741.71
	Precision weighted average	7.07			
	Unweighted average	158.86			

Table 10: Sevilleta and Bosque del Apache NWRs Meta-Analysis Benefit Transfer Monte Carlo quantiles

	Sevilleta and Bosque del Apache NWRs	Predicted per-acre value, $e^{E[X\beta]}$	Simulated distribution of expected per-acre value		
			5th percentile	50th percentile	95th percentile
Flood Control	Woodward and Wui	1487.34	4.39	1483.62	501931.60
	Brander, Florax, and Vermaat	6.07	0.0000041	6.00	8991305.62
	Ghermandi et al.	18.44	0.20	18.51	1666.25
	Precision weighted average	18.44			
	Unweighted average	503.95			
Water Quality	Woodward and Wui	1577.75	4.71	1572.97	531797.40
	Brander, Florax, and Vermaat	9.91	0.0000065	9.77	15247340.97
	Ghermandi et al.	24.76	0.28	24.89	2227.39
	Precision weighted average	24.76			
	Unweighted average	537.47			

Table 11: Flood control Monte Carlo mean and variance

Flood control Refuge	**Mean, $E[e^{X\beta}]$**		**$Var(e^{X\beta})$**	
Arrowwood NWR	Woodward	2.98E+05	Woodward	1.02E+08
	Brander	2.57E+11	Brander	1.93E+22
	Ghermandi	638.5	Ghermandi	133.81
Blackwater NWR	Woodward	2.47E+05	Woodward	1.31E+08
	Brander	3.26E+11	Brander	5.39E+22
	Ghermandi	2095.9	Ghermandi	1776.4
Okefenokee NWR	Woodward	1.52E+05	Woodward	7.13E+07
	Brander	1.17E+10	Brander	1.18E+19
	Ghermandi	221.69	Ghermandi	20.146
Sevilleta and Bosque del Apache NWRs	Woodward	2.92E+05	Woodward	1.01E+08
	Brander	1.80E+10	Brander	2.19E+19
	Ghermandi	645.56	Ghermandi	113.82

Table 12: Water quality Monte Carlo mean and variance

Water quality				
Refuge	**Mean, $E[e^{X\beta}]$**		**$Var(e^{X\beta})$**	
Arrowwood NWR	Woodward	3.23E+05	Woodward	1.62E+08
	Brander	5.93E+11	Brander	9.73E+22
	Ghermandi	848.31	Ghermandi	208.96
Blackwater NWR	Woodward	2.60E+05	Woodward	1.10E+08
	Brander	7.19E+11	Brander	2.60E+23
	Ghermandi	2785.6	Ghermandi	3138.4
Okefenokee NWR	Woodward	1.64E+05	Woodward	9.41E+07
	Brander	2.32E+10	Brander	5.37E+19
	Ghermandi	294.1	Ghermandi	29.295
Sevilleta and Bosque del Apache NWRs	Woodward	3.17E+05	Woodward	1.59E+08
	Brander	3.80E+10	Brander	1.66E+20
	Ghermandi	859.97	Ghermandi	190.27

Table 13: Estimated WTP per average acre per year and for refuge wetlands per year

Meta-analysis	NWR site	Service valued	Median value per average acre	NWI wetland acres	Median value per refuge
Woodwardand Wui (2001) Meta-analysis	Arrowwood	Flood control	$1,533	*4,595*	$7,044,000
		Water quality	$1,626	4,595	$7,472,000
	Blackwater	Flood control	$845	24,502	$20,700,000
		Water quality	$896	24,502	$21,959,000
	Okefenokee	Flood control	$435	375,778	$163,462,000
		Water quality	$461	375,778	$173,393,000
	Sevilleta and Bosque	Flood control	$1,487	5,106	$7,594,000
		Water quality	$1,578	5,106	$8,056,000
Brander, Florax, and Vermaat (2006) Meta-analysis	Arrowwood	Flood control	$2.65	*4,595*	$12,200
		Water quality	$4.33	4,595	$19,900
	Blackwater	Flood control	$13.91	24,502	$341,000
		Water quality	$22.70	24,502	$556,000
	Okefenokee	Flood control	$4.96	375,778	$1,863,000
		Water quality	$8.09	375,778	$3,042,000
	Sevilleta	Flood control	$6.07	5,106	$31,000
		Water quality	$9.91	5,106	$50,600
Ghermandi et al. (2010) Meta-analysis	Arrowwood	Flood control	$17.48	*4,595*	$80,300
		Water quality	$23.48	4,595	$107,900
	Blackwater	Flood control	$46.16	24,502	$1,131,000
		Water quality	$62.00	24,502	$1,519,000
	Okefenokee	Flood control	$5.27	375,778	$1,979,000
		Water quality	$7.07	375,778	$2,659,000
	Sevilleta	Flood control	$18.44	5,106	$94,100
		Water quality	$24.76	5,106	$126,400

CHAPTER 4

A NOVEL META-ANALYSIS

This chapter details the procedure for developing and modeling a meta-analysis database of wetland ecosystem service valuation studies. The meta-analysis database is modeled for benefit transfer applications and estimated with a conventional, multivariate ordinary least squares regression. I first discuss the data gathering process and describe some of the challenges to incorporating diverse valuation studies into a single database and estimation framework. Next the specification of the OLS model is described followed by results and discussion of those results.

The wetland valuation database follows the broad wetland valuation meta-analysis models considered in Chapter 3.The analysis in Chapter 3 illustrates important reasons for conducting a novel meta-analysis. A variety of assumptions are required to code a dataset based on heterogeneous primary valuation studies, and only a large database can document these assumptions. Ultimately, the experience of developing a meta-analysis database is useful for understanding the limitations of the models used to analyze the database. Similarly, an understanding of the limitations of forecasts for used for benefit transfer applications requires familiarity with the underlying data set. In the chapter below, I communicate some of the challenges of the meta-analysis modeling of ecosystem service valuation studies, but inevitably this communication is only a summary.

Data

While theory offers little formal guidance for the meta-analysis practitioner specifying the model to be estimated, existing meta-analysis studies provide some guidance concerning functional form and relevant explanatory variables. Once the meta-analysis practitioner has assembled a list of explanatory variables to include in the model, obtaining the values of each variable for each observation is the next task. Challenges can potentially arise when attempting to find measurements of both the dependent variable and the explanatory variables. The paragraphs below describe the data gathering process necessary for measuring the values of variables associated with primary valuation studies and how some of the inevitable problems associated with combining diverse studies are addressed. Because the data gathering process occurs first, some variables are included in the database that are not used in the final model; these variables may be useful for future modeling efforts.

Many existing wetland ecosystem service meta-analysis studies specify the dependent variable in terms of WTP per acre/hectare or WTP per person, and only a single non-peer-reviewed wetland meta-analysis employs both (Borisova-Kidder 2006). For modeling purposes, observations that are retained in the final data set must share a common normalization or units of the dependent variable. These units are typically WTP per unit surface area or per person. Careful consideration and additional information is important when combining primary valuation results with different units and different welfare measures. The main assumption of the meta-analyst is that explanatory variables can be specified that control for these many sources of variation.

In a search of the literature, I obtain and examine all studies in the reference lists of the wetland meta-analysis studies in Table 1. To provide a comparison with these studies and to provide a means for assessing correspondence across services, primary valuation studies included in this initial census that do not value water quality enhancements or flood control/storm protection are also retained. In order to include additional relevant wetland valuation studies, the following valuation databases were searched for domestic wetland valuation studies: Environment Canada’s Environmental Valuation Reference Inventory, the Marine Ecosystem Services Partnership’s Valuation Library, the Gulf of Mexico Ecosystem Services Valuation Database, and Google Scholar. I also reviewed the reference lists of primary valuation studies as they were added to the database. This census of the literature ultimately yielded as many as 350 possible value estimates from 277 studies.

From the initial census of 277 studies, several criteria were applied to filter the census into a final dataset for modeling. Beginning with domestic wetland studies, this filtering process reduces the influence of data found in multiple valuation studies, facilitates geospatial modeling, and ensures that all explanatory variable values exist for each observation. First, studies that did not use original data were eliminated from the meta-analysis dataset. This included studies that reused primary valuation data and benefit transfer valuation studies. Next, I removed studies that provided insufficient information about the study site to allow for geospatial analysis. The basic requirement for geospatial analysis is that the primary valuation study clearly identifies the wetlands being valued so that they can be identified on a map. I searched for related research reports and working papers that might provide geospatial details missing from publications focusing on econometric contributions. Ultimately, this filtering process resulted in 26 useable studies for the meta-analysis. These 26 studies included 82 observations associated with 53 unique georeferenced sites.

Following similar wetland ecosystem service valuation meta-analysis studies, I include a variety of economic valuation approaches as a component of this study's econometric identification strategy. The inclusion of multiple primary valuation approaches allows for a larger sample with more variation in explanatory variables. A summary of the 82 observations used in the meta-analysis regressions is provided in Table 14.The first column of the table contains variable names, with the dependent variable (2010 US dollars per 1000 acres per person per year) on the first row below the header. The second column is the sample mean of the 82 observations and the third column is the variance of the sample mean. All method-related variables are binary or dummy variables as is the variable, "coastal" for sites that are on the coast of an ocean, which applies to Blackwater NWR in my modeling application.

The most frequently used approach for valuing wetland ecosystem services in the novel data set is the stated preference approach. The Contingent Valuation Method provides 46 observations, and 14 observations are from choice experiments. Both of the stated preference methods provide observations of

WTP per person, typically as Hicksian Compensating Variation. Most of the remaining studies utilize the travel cost method, which provides 15 observations.

The remaining observations are from studies that utilize economic valuation methods that are generally considered to be less theoretically appropriate due to the burden of stronger assumptions. I lump together into a single omitted dummy variable studies that utilize damage avoidance and replacement cost (of capital). The inclusion of these studies, provides 7 additional observations, in order to obtain better model flood control/storm protection and water quality provisioning services.

Understanding the welfare effect of a wetland landscape requires value estimates that can be aggregated across the landscape and user population. In primary valuation studies the measured WTP of a representative agent is often aggregated to the relevant user population, typically through multiplication (e.g., Bergstrom et al. 1990). More advanced approaches may segment the population by observable characteristics (e.g., Cooper and Loomis 1993; Bishop et al. 2000) or may include information to estimate distance decay (Sutherland and Walsh 1985). Distance decay is the intuitive concept that people further from a site will benefit less from the site than people who are closer. However, because not all studies contain estimates of aggregate WTP, this process requires additional information from the meta-analysis practitioner. Similar problems arise with the Travel Cost Method which may report WTP normalized by person, trip, or household.

When additional information is required for estimating aggregate WTP associated with an observation, the meta-analysis practitioner typically must make a decision about how to proceed, potentially introducing error into the model. Some primary valuation studies may specify the geographic extent from which the sample was obtained, often delimited by political boundaries (e.g., Cooper and Loomis 1993; Phaneuf and Herriges 1999), but the study may not report relevant socio-economic data on the population of the area. For example, the local population count and average income and are not reported in many valuation studies. For instances such as these, US Census data provide population and income information for counties and states associated with the user population. When socio-economic data are reported, I verify these numbers with US Census data.

One of the most important aspects of the data gathering process that may require supplementary information beyond the primary valuation study is the identification of the geographic extent of the wetland that is valued. Because most existing wetland meta-analysis studies include a surface area regressor, allowing for non-constant returns to wetland area implies that both aggregate values and values-per-acre may be misestimated if wetland extent is measured with error. Identifying the extent of the wetland valued in an observation in the data set also allows for more complex geospatial analysis of the site's geographic context. Thus I impose the requirement that the study must provide sufficient information to identify the geographic extent of the studied wetland. I also eliminate from the database those studies that provide a value for extremely large wetland extents, such as the entire country (e.g., Bergstrom and Cordell 1991).

The additional analysis and decisions that the meta-analysis practitioner needs to make in order to estimate the extent of valued wetlands can be usefully illustrated by example. In Breaux, Farber, and Day (1995) a wastewater treatment wetland in Thibodaux, La is identified but with too little information to identify the site on a map. By searching the internet for documents relating to the "State Department of Environmental Quality", "Thibodaux, Louisiana", and terms relating to wastewater treatment and discharge, I found a document describing the site. The site in Twilley and Boustany's (1990) report matches closely the description in Breaux, Farber, and Day (1995), including the extent of treatment wetlands (230 hectares vs. 570 acres, respectively) and both describe a two ridge system with a drainage canal. Additionally, both reports cite locally relevant studies by William H. Conner and John W. Day (e.g., Conner, Day, and Bergeron 1989). Other similar documents confirming the site were also found. A document by Twilley and Boustany (1990) provides additional information, allowing for identification of the site in the EPA's Water Quality Assessment Status reports (http://watersgeo.epa.gov/mwm/?layer=305B&feature=LA120207_00&extraLayers=null), providing sufficient information for identifying the boundaries of the site in Arcgis. I use this information to construct a polygon matching the boundaries of the site, allowing for analysis of the surrounding population and geography.

While the Breaux, Farber, and Day (1995) study values a small geographically specific wetland, Bergstrom, Stoll Titre, and Wright (1990) value a substantially larger wetland system, estimated at 3.25 million acres by the authors. Their Figure 3, a map of the counties in the study region, identifies the appropriate Louisiana counties. Because Bergstrom et al. (1990) attribute all consumer surplus in their contingent valuation study to wetlands in the selected counties, I retain all wetlands therein as the scope of valuation. The procedure for identifying coastal wetlands valued in Costanza, Farber, and Maxwell (1989) was similar. In their study a section includes estimates of the value of hurricane protection services provided by Terrebonne Parish coastal wetlands. I define their site first by selecting wetlands in Terrebonne Parish and then selecting by eye the subset of these wetlands that are between populated areas and the Gulf of Mexico.

The question of how big the wetland study site is can potentially have an unclear answer. Stated preference studies evaluate willingness to pay to protect or prevent the loss of a precisely defined wetland area (i.e., indicated on a map or with a precise description) (e.g., Sutherland and Walsh 1985; Sanders, Walsh, and McKean 1991), of a generic, and locally relevant wetland (e.g., Johnson and Linder 1986; Blomquist and Whitehead 1998; Bauer, Cyr, and Swallow 2004), or of a certain portion of a larger nearby wetland area(e.g., Beran 1995; Bishop et al. 2000; Whitehead et al. 2009; Petrolia and Kim 2011). Even for sites that value wetlands of a well-defined extent, the wetland ecosystem around or near the site may be substantially larger. Attributing the average per-acre willingness to pay for a smaller subset of wetland acres to all adjacent wetland acres may lead to a large upward bias in the aggregate welfare measure. Accordingly, when applicable, I specify the wetland surface area variable as the value discussed in the primary valuation survey. As discussed below, the accounting of substitute wetlands in a radius around the study site as an explanatory variable provides a control for changes in WTP that are related to the broader wetland landscape.

Beyond estimating the size of the wetlands being valued, a further conceptual issue that arises when identifying wetland acreage from a primary valuation study occurs when a study values a hypothetical or generic rather than actual wetland. Johnston et al. (2002) and Bauer, Cyr, and Swallow (2004) are examples of valuation studies where a hypothetical or generic site is valued. Both of these studies, however, indicate a context that motivated the research; I use this context as the actual site of valuation in order to be able to obtain information about the geographic context of the wetland.

The relatively recent wetland meta-analysis studies of Brander, Florax, and Vermaat (2006) and Ghermandi et al. (2010) include dummy variables distinguishing wetland type. I follow this approach, but estimate continuous variables representing the proportion of wetlands at each site in each subsystem, as defined by Cowardin et al. (1979). Because many wetland valuation studies consider wetlands that include a variety of wetland subsystems, this flexible approach can capture more variation in sites and is less prone to measurement error than a binary approach at the expense of requiring more geospatial analysis. However, because I use other spatial variables, there is little difficulty in measuring the wetland subsystem proportions beyond identifying the boundaries of each site.

With a well-defined digital map of each wetland, one can assess more complex geospatial variables. In order to control for the availability of substitute wetland sites, I follow Ghermandi et al. (2010) who include a variable for the total area of wetlands in a 50km radius around the site. A non-static radius for counting nearby wetlands provides a more flexible means of accounting for the surrounding landscape. Valuation studies that value geographically extensive wetlands, such as all the wetlands in a particular state, are likely to be associated with a wider scope for substitution than studies that value small wetlands. Because of this concern I develop a more flexible method, discussed below, for measuring substitutes that varies with wetland surface area. I use a similar approach to count local populations. NASA-SEDAC's Gridded Population of the World V3 (http://sedac.ciesin.columbia.edu/data/collection/gpw-v3) data provides a means to count the local population in the same area as substitute wetlands. The inclusion of these local population and local substitute wetland variables in the data set serves to approximate the demand and scarcity of wetland services associated with each site.

Because the model requires wetland surface area for all observations, this information is used to define the substitute wetland/local population radius as a function of each study site's surface area. Due to the range of wetland sizes in the primary valuation database, a number of functional forms led to overly wide variations in the radius. This observation led me to experiment with several monotonic transformations of the surface area for each site to establish an intuitively reasonable radius for counting substitute wetlands and the local population. The basic objective was to find a functional form that was at least a mile for small wetlands, but which did not extend beyond the North American continent for very large wetlands. After experimentation with several functional forms, I chose a functional form that takes the cube root of wetland acreage as the radius, but with the units changed to miles using a non-standard 1:1 conversion. Notably, I did not choose a conversion based on geometric considerations, as I have no expectation that Euclidean geometric conversions such as from linear to surface area units will better approximate the relationship between populations and wetlands than the intuitive approach I followed. To account for land cover change, the NLCD data set that is closest in time to the year of each primary study's data acquisition defines substitute wetlands. An important limitation in the accounting of substitute sites is that different wetland types are not distinguished. For example, a wetland study site that consists only of brackish wetlands for which freshwater wetlands may be a poor substitute has included in the count of substitutes both fresh water and brackish wetlands.

Method

Broad meta-analysis models that include observations associated with diverse socio-economic conditions, services, methods, and ecosystems require careful consideration of the consequence of parsimonious variable specification. Yet a fully specified model with all relevant interaction effects among regressors may fail the full-rank condition necessary for computation of the OLS estimator. The meta-analysis practitioner faces the choice of whether or not to use a sample with a broad scope of included observations that may require estimation of a large number of parameters in order to convincingly control for sources of bias.

Results

The results of the OLS regression can be found below in Table 15. The table reports Huber-White or heteroskedasticity robust standard errors, t-statistics, and the p-values associated with the t-statistic for each parameter. The parameters estimated for both water quality and flood control are significant at the 10% level or better. The water quality parameter is significant at the 5% level; the flood control/storm protection parameter just barely misses that mark, suggesting that the parameter estimates are reasonably precise.

The parameter estimate for the number of acres valued is negative and significant at better than the 10% level with robust standard errors. The parameter for population is similar in magnitude to the parameter for acres, but the p-value is less than 1%. Because the dependent variable is normalized by acres and population, the interpretation is that a 1% increase in one of these variables leads to about a 0.5% decrease in WTP/acre/person, which suggests diminishing returns to expanding the scope of acres valued or the scope of the population for aggregation. The variable, *joint*, which indicates how many services were jointly valued is strongly significant and indicates that adding additional services to a valuation study results in a reduction of the estimated value relative to a model where the additional services were valued separately; this result is surprising and merits future research. The results of the OLS forecasts are discussed in greater detail along with the results of the PLWLS estimator in the following chapters.

Table 14: Meta-analysis Data Summary

Variable	Mean	Variance
n=82		
Dependent		
Value/1000ac/pop†	24.29416996	13584.72966
Study/Site		
Acres valued†	154934.3668	2.29501E+11
Population†	3531250.744	5.66455E+13
Method-related (1/0)		
Valuation Approach		
Revealed preference	0.18293	0.15131
Stated preference	0.73171	0.19874
Joint valuation (1,2,3)	1.1098	0.17299
Service Valued		
Water quality	0.12195	0.1084
Flood protection	0.060976	0.057964
Total value	0.5	0.25309
Recreation, general	0.12195	0.1084
Habitat	0.085366	0.079042
Recreation, fishing	0.02439	0.024089
Recreation, hunting	0.15854	0.13505
Interaction- use	0.073171	0.068654
Interaction- passive	0.20732	0.16637
Context		
GDP(state)†	31205.5105	37717998.95
Coastal (1/0)	0.12195	0.1084
GDP*local_pop†	25825016098	2.01947E+21
Local_pop:local_wet†	47.06399473	0.057448
† mean and variance in levels, but variable used in log form		

Table 15: OLS Meta-analysis Regression Model Results

Variable	**OLS Coefficient**	**standard error**	**T-statistic**	**P-value**
Intercept	24.147	19.863	1.216	0.23
Study/Site				
Acres valued†	-0.583*	0.306*	-1.904	0.06
Population†	-0.41***	0.148***	-2.761	0.01
Method-related (1/0)				
Valuation Approach				
Revealed preference	-4.401*	2.319*	-1.898	0.06
Stated preference	0.864	1.305	0.662	0.51
Joint valuation (1,2,3)	-5.977***	1.435***	-4.164	0
Service Valued				
Water quality	5.969**	2.864**	2.084	0.04
Flood protection	5.438*	2.861*	1.9	0.06
Total value	6.534**	2.836**	2.304	0.02
Recreation, general	10.667***	2.627***	4.061	0
Habitat	6.564**	3.093**	2.122	0.04
Recreation, fishing	5.287**	2.178**	2.428	0.02
Recreation, hunting	5.011*	2.628*	1.907	0.06
Interaction_use	-1.211**	0.562**	-2.153	0.04
Interaction_passive	-0.705	0.56	-1.258	0.21
Context				
GDP(state)†	-2.681	2.076	-1.291	0.2
Coastal (1/0)	1.214**	0.566**	2.144	0.04
GDP*local_pop†	0.038	0.029	1.309	0.2
Local_pop:local_wet†	2.394**	1.124**	2.13	0.04
$R^2 = 0.81$				
Significance levels: ***<0.01, **<0.05, *<0.1				

CHAPTER 5

THE PARAMETRIC LOCALLY WEIGHTED LEAST SQUARES ESTIMATOR

The parametric locally weighted least squares estimator developed below has at its core a generalized least squares-like approach to efficient estimation of regression coefficients. GLS is a weighted least squares estimator that is widely known to produce an estimate of regression parameters that are more efficient than the conventional OLS estimator when the correct weight matrix is used (Hayashi 2000, 137). The weight matrix is usually a diagonal matrix where each element is the respective observation's known inverse error variance. However the true weight matrix is not known for empirical applications and must be estimated. The purpose of the estimator developed below is a new approach to the estimation of the appropriate weight matrix for any benefit transfer of interest.

In the meta-analysis benefit transfer literature, high correspondence between a sample of study sites and a policy site is often implemented in a model by *ad hoc* resampling from an initial census of the literature. This re-sampling procedure uses intuition based rules that implicitly weight observations with 1's and 0's. The selection process is based on whether a meta-analyst thinks the selection rules will result in a sufficiently robust dataset to allow regression modeling that also does not suffer from poor correspondence. Importantly, the effects of the resampling process on the validity and precision of benefit transfers has not been explored in the literature. The benefits of a more rigorous, systematic approach to resampling motivate the development of the PLWLS estimator.

The main alternative to resampling from a census of the literature when developing a meta-analysis model is to model the census of available literature with a single equation. With this approach, the modeler accepts an imprecise model in exchange for avoiding the introduction of bias through resampling. The use of the broad meta-analysis models described in Chapter 2 for benefit transfer would be an example of an imprecise estimate, but by avoiding *ad hoc* resampling, a statistical description of the behavior of the broader model is still available.

In Figure 13, I provide a conceptualized illustration of the trade-offs between the benefits of a larger, random sample and the benefits of high correspondence. Points towards the northeast corner of the space defined by the vertical sample size axis and horizontal correspondence axis are associated with a better, more ideal model. The main idea behind this figure is that the best modeling strategies retain both the benefits of the robust information associated with a large sample estimated with a systematic model and the benefits of precise benefit transfers associated with a sample containing observations with high correspondence to the policy site. The two orange circles are indicative of the two groups of meta-analysis studies discussed in Chapter 2. The large sample size meta-analysis studies that focus on hypothesis testing are represented by the orange circle on the vertical axis, and the small sample, high correspondence benefit transfer targeted meta-analysis studies are represented by the orange circle on the horizontal axis. My hypothesis is that the yellow circle represents benefit transfers implemented with PLWLS forecasts. The three circles all lie on a hypothesized correspondence/sample-size frontier, but the yellow point is preferable because it combines most of the advantages with few of the drawbacks of the two orange points.

One of the main advantages of estimating a regression centered on each site in the sample is that doing so allows for variation in parameters as a site's *correspondence attributes* change relative to the sample. The typical regression models used in meta-analysis studies of ecosystem services do not estimate the wide variety of interaction effects that may exist among variables. The meta-analysis studies of wetland ecosystem services discussed in Chapter 2, with the exception of Ghermandi et al. (2010) do not discuss in much or any depth the possibility that these interaction effects exist. Because the sample is complex, restricting the partial effects of regressors to be constant across all meta-analysis benefit transfer applications may lead to bias and imprecision. The method this chapter develops is intended to improve on conventional approaches by allowing any observation to potentially have unique regression parameters, a more flexible specification than the single equation OLS model.

The exposition of the method begins with a description of the ideas behind both local and weighted regressions and then identifies how the new approach differs from common single equation WLS estimators. Next, I describe a unique calibration and forecast approach that serves to identify *correspondence parameters*, which determine the weight given to a particular study site in the equation for any policy site. These weights are calculated using a multivariate measure of the “distance” between each study site and each policy site in an equation specific to the policy site. Next the procedure for using estimated correspondence parameters to forecast benefits for out-of-sample policy sites is described. To further explain the PLWLS estimator, I provide step-by-step instructions for the calibration stage and the forecast stage.

The PLWLS estimator is then applied to the novel data set from Chapter 4, and the results are compared to the OLS model and an alternative specification of the PLWLS estimator. The purpose of the estimator developed in this chapter is twofold. First, the estimator offers a quantitative means for formalizing the intuitive notion of correspondence or site similarity (Johnston 2007), and second, the estimator allows for a potentially unique meta-regression equation for benefit transfer at any policy site with only a single model.

Method

The basis of the PLWLS estimator is a regression equation that can be estimated with OLS; this feature facilitates a direct comparison of the two estimators. Because the PLWLS method formalizes the theoretical notion that high correspondence among observations leads to reduced transfer error, it is important to fix the basic idea of the approach before exploring the detailed steps necessary for estimation. Ultimately, the goal is to estimate an accurate regression for each site in the sample, but without relying on non-random resampling performed by an analyst.

Every site in the sample receives two indices, i and j, where the former indicates that a site is the object of a benefit transfer and all other sites (indexed by j) are the sources of data for estimating a benefit transfer function specific to site i. The ultimate objective of the approach is the development of an efficient estimator of B_i which is intended to be used for forecasting benefits for site i. Consider the following meta-regression equation for site i.

$$y_j = x_j B_i + e_{ij} \tag{8}$$

where y_j is a scalar variable representing a measured (by a primary valuation study) value with units in WTP per acre per person. The variable x_j is a row-vector of k explanatory variables for site j. The parameter B_i is the k element population parameter column-vector for site i, and e_{ij} is a scalar, zero mean stochastic population error term associated with the data x_j and y_j and the parameter vector, B_i. The classical regression approach leads to the estimate of a value of B_i that minimizes the sum of squared errors across the sample of n predictions of the dependent variable, y. The parameter vector for observation i, B_i, is indexed by i to indicate that this parameter is optimized to efficiently forecast the dependent variable for a particular site, i. As one cannot observe B_i it must be estimated; estimates of the population parameter B_i are represented as $\tilde{B}_i$. A tilde over a variable indicates that the variable has been estimated by PLWLS. I am interested in precise estimates of B_i, that is where e_{ij}^2 can be expected to be as low as feasible for the available data. Generally, cases where $i = j$ are of interest, indicating that the particular choice of B_i is intended for the data at hand, while cases where $i \neq j$ are typically not of interest except during calibration. In the typical regression context, B_i is the same for all observations, assuming a common sample and regression specification are used for each observation. This similarity in parameter estimates across observations is relaxed with the PLWLS estimator.

The most straightforward and familiar estimator of B_i for data y_i and x_i is the OLS estimator, $\hat{B}_i$,

$$\hat{B}_i = (\boldsymbol{X}'_{(i)}\boldsymbol{X}_{(i)})^{-1}\boldsymbol{X}'_{(i)}Y_{(i)}. \tag{9}$$

Here $\boldsymbol{X}_{(i)}$ is a $(n_i \times k)$ matrix which contains ni rows of the k element data row-vectors x_j. Analogously, $Y_{(i)}$ is a $(n_i \times 1)$ column vector of scalar values of y_j. The subscript, j may take on values from 1 to n_i. The subscript, (i), indicates that a particular data matrix is intended for calculating $\hat{y}_i$, that is an estimate of the originally measured value of y_i. For benefit transfer targeted meta-analysis models, a common procedure is to choose or weight by some means the rows of $\boldsymbol{X}_{(i)}$ and $Y_{(i)}$ such that the benefit transfer for site i, an estimate of y_i, is precise relative to the available information. The motivation for down-weighting or eliminating an observation is typically that the analyst thinks that observation is relatively uninformative about the site of interest due, for example, to an analyst's qualitative expectations of poor study/site correspondence. The benefit transfer targeted meta-analysis studies discussed in Chapter 2 accomplish this weighting implicitly by discarding observations thought to offer little information for the benefit transfer application of interest. The purpose of the PLWLS estimator is to model this selection procedure empirically and explicitly.

In order to estimate a weighted regression, I modify equation (9) by multiplying the data vectors by a diagonal vector of estimated weights to obtain the WLS equation centered on observation i, or local WLS regression,

$$\tilde{B}_i = (\boldsymbol{X}_{(i)}{}'\widetilde{\mathbf{W}}_i\boldsymbol{X}_{(i)})^{-1}\boldsymbol{X}_{(i)}{}'\widetilde{\mathbf{W}}_i Y_{(i)}. \tag{10}$$

When weights are introduced, the (i) subscript on the data matrix $\boldsymbol{X}$ and vector Y indicates that the i^{th} row is dropped from the matrix and all other rows are retained, so for all i, $n_i = n - 1$. The fundamental intuition behind a weighted regression is that certain observations contain less information about the underlying population from which the sample was drawn. By correctly reducing the weight of those less informative observations, the relative information content across all observations is preserved. In contrast, if one knew the appropriate weights but did not use them, the more informative observations would be effectively down-weighted relative to the information they contain about the population; in other words, if one were to ignore these weights then useful information would be watered down with extraneous random variability, resulting in an imprecise model..

In order to quantify the idea of study and site correspondence, I assume that the optimal weights in each regression centered about observation i are related to sample errors by the following equation, known as the *correspondence equation*,

$$\tilde{e}_{ij}^2 = \widetilde{w}_{ij}^{-1} + \tilde{u}_{ij}. \tag{11}$$

Where $\widetilde{w}_{ij}$ is the square of the estimated data weight applied in a WLS model for observation j in the regression centered on observation i; $\widetilde{w}_{ij}$ is also the diagonal element of $\widetilde{\mathbf{W}}_i$ that corresponds to observation j. The term $\tilde{e}_{ij}$ is the residual from the j^{th} observation in the i^{th} centered regression equation. The term $\tilde{u}_{ij}$ is assumed to be a mean zero stochastic residual term. All of the terms in this equation are estimates or sample counterparts (as denoted by the tilde) to what I assume to be the true underlying local residuals, weights, and correspondence residuals. The aim of the PLWLS estimator is to use information about each site to model the relationship in equation (11) in order to estimate local WLS weighted regressions for out-of-sample sites.

In order to estimate equation (11) I assume that the optimal w_{ij} is a function of *correspondence attributes* at each site and true global *correspondence parameters*, $\boldsymbol{\delta} = [\delta_1, \delta_2, \dots, \delta_H]'$ that are shared across all sites and all centered models. Because these *correspondence parameters* are unobservable, they must be estimated. The objective function for estimating the *correspondence parameters* is as follows,

$$\min_{\{\boldsymbol{\delta}\in\mathbb{R}^{\mathrm{H}}:\boldsymbol{\delta}\geq 0\}} \sum_{i=1}^{n} \sum_{\substack{j=1 \\ j\neq i}}^{n} (\widetilde{\mathrm{w}}_{ij}\tilde{u}_{ij}^2)\,. \tag{12}$$

The tildes indicate that each variable has been estimated by PLWLS. The outer sum indexes the n centered regression equations where observation i is treated as a centered or policy site. The inner sum indexes the study sites, indexed by j, that serve as information for modeling the centered or policy site i. In equation (12), the use of the estimated *correspondence weights*, $\widetilde{\mathrm{w}}_{ij}$, leads the optimization routine to favor smaller values of $\tilde{u}_{ij}^2$ when observation j (serving as a study site in this context) in the equation centered on observation i has high correspondence (and thus a large weight) with observation i (serving as the policy site in this context). Equivalently, when correspondence is high, the weight visible in (12) leads to a higher penalty when the squared regression residual is far from the estimated WLS weight. I chose this additional weight in the objective function because large values of u_{ij}^2 are less of a concern when estimated by a centered equation that is expected to be imprecise due to poor correspondence between the centered observation and the observation indexed by j.

In order to estimate equation (11) using the objective function in equation (12), I assume that the true correspondence weights, w_{ij}, are a function of $h = 1, \dots, H$ scalar *correspondence attributes*, a_{hi} and a_{hj}, at sites i and j, and H estimated *correspondence parameters*, $\tilde{\delta}_h$, common to all observations. I also use bold to denote the H element vector versions of correspondence attributes, $\boldsymbol{a}_i$ or $\boldsymbol{a}_j$, and estimated correspondence parameters, $\widetilde{\boldsymbol{\delta}}$. Specifically, when considering the weight of a set of observations indexed by j in the population centered about site i, I am interested in the estimated weight matrix, $\widetilde{\boldsymbol{W}}_i = g(\boldsymbol{a}_i, \dots, \boldsymbol{a}_j; \widetilde{\boldsymbol{\delta}}, i)$, where $g(.)$ denotes the *correspondence weight equation* and $\widetilde{\boldsymbol{\delta}}$ is a vector of *correspondence parameters* that will be estimated during the calibration stage.

I specify the functional form of the *correspondence weight equation*, $g(.)$ as the negative exponential functional form,

$$g(\boldsymbol{a}_1, \dots, \boldsymbol{a}_n; \widetilde{\boldsymbol{\delta}}, i) =$$

$$diag\left\{exp\left(-\begin{bmatrix} |\mathrm{a}_{1i}-\mathrm{a}_{11}| & |\mathrm{a}_{2i}-\mathrm{a}_{21}| & \dots & |\mathrm{a}_{Hi}-\mathrm{a}_{H1}| \\ |\mathrm{a}_{1i}-\mathrm{a}_{12}| & |\mathrm{a}_{2i}-\mathrm{a}_{22}| & \dots & |\mathrm{a}_{Hi}-\mathrm{a}_{H2}| \\ \vdots & \vdots & & \vdots \\ |\mathrm{a}_{1i}-\mathrm{a}_{1(\mathrm{i}-1)}| & |\mathrm{a}_{2i}-\mathrm{a}_{2(\mathrm{i}-1)}| & \ddots & |\mathrm{a}_{1i}-\mathrm{a}_{\mathrm{H}(\mathrm{i}-1)}| \\ |\mathrm{a}_{1i}-\mathrm{a}_{1(\mathrm{i}+1)}| & |\mathrm{a}_{2i}-\mathrm{a}_{2(\mathrm{i}+1)}| & & |\mathrm{a}_{1i}-\mathrm{a}_{\mathrm{H}(\mathrm{i}+1)}| \\ \vdots & \vdots & & \vdots \\ |\mathrm{a}_{1i}-\mathrm{a}_{1n}| & |\mathrm{a}_{2i}-\mathrm{a}_{2n}| & \dots & |\mathrm{a}_{Hi}-\mathrm{a}_{Hn}| \end{bmatrix}\begin{bmatrix}\tilde{\delta}_1 \\ \tilde{\delta}_2 \\ \vdots \\ \tilde{\delta}_H\end{bmatrix}\right)\right\} \tag{13}$$

$$= \widetilde{\mathbf{W}}_{\mathrm{i}}.$$

For each dimension of correspondence, indexed by h, equation (13) takes the absolute difference between the hth *correspondence attribute* at site i (a_{hi}) and site j (a_{hj}) multiplied by an estimate of the correspondence parameter, $\tilde{\delta}_h$. The operator $diag\{\}$ converts the column vector within to a diagonal matrix. The function, $g(.)$, notably does not include a term where observation i is compared with itself. Because I do not know the true values for δ_h, I use equation (12) to estimate them.

For the empirical application of the PLWLS model, the data and regression specification in Chapter 4 are retained. The only additional specification required for PLWLS relates to the choice of variables that serve as *correspondence attributes* appearing in equation (13). I choose for the empirical application to include in log form the following continuous variables as correspondence variables: GDP per capita, distance between sites in 1000's of kilometers, a count of the population of beneficiaries (technically, a count variable), and acres of wetland valued. I also include several binary variables as correspondence variables, which includes a flood control/storm protection service indicator variable, a water quality service indicator variable, and a variable to indicate if a site is coastal. All of these variables are also included as conventional explanatory variables in the main regression equation.

The functional form in equation (13) was chosen because it satisfies two important theoretical characteristics of correspondence. First, when the difference between two correspondence attributes, $d_{ijh} = |\mathrm{a}_{hi} - \mathrm{a}_{hj}|$, grows, then correspondence between observations i and j is lower. When correspondence is lower, the weight that observation j receives in the regression centered on observation i is decreased to reflect the reduced information that the observations provide about each other. Second, when a particular estimated correspondence parameter, $\tilde{\delta}_h$, grows, the impact of a correspondence difference, $d_{hij} = |\mathrm{a}_{hi} - \mathrm{a}_{hj}|$, is larger, implying reduced correspondence or a smaller weight. This relationship is expressed in the following partial derivatives,

$$\frac{\partial(\tilde{\mathrm{w}}_{\mathrm{ij}})}{\partial(d_{hij})} = \frac{\partial\left(\widetilde{\mathbf{W}}_{\mathrm{ijj}}\right)}{\partial(d_{hij})} \leq 0$$

$$(14)$$

$$\frac{\partial(\tilde{\mathrm{w}}_{\mathrm{ij}})}{\partial(\tilde{\delta}_h)} = \frac{\partial\left(\widetilde{\mathbf{W}}_{\mathrm{ijj}}\right)}{\partial(\tilde{\delta}_h)} \leq 0.$$

With the model fully specified, the objective function in equation (12) can be better understood by observing that the estimated correspondence parameters, $\tilde{\delta}_h$, appear in $\tilde{\mathrm{w}}_{ij}$ and also in $\tilde{u}_{ij}^2$. This dependence on estimates of $\tilde{\delta}$ is also communicated with the tilde (~) over the residuals and weights and subsequent variables that are also a function of $\tilde{\delta}_h$. Substitution and simplification reveals the complex, non-linear nature of the objective function,

$$\min_{\{\boldsymbol{\delta}\in\mathbb{R}^{\mathrm{H}}:\boldsymbol{\delta}\geq 0\}} \sum_{i=1}^{n} \sum_{\substack{j=1 \\ j\neq i}}^{n} \left(exp^{-\sum_{h=1}^{H}|a_{hi}-a_{hj}|\tilde{\delta}_h} \left(\mathrm{y_j} - \mathrm{X_j}\left[\left(\boldsymbol{X}_{(i)}{}'g(a_1,\dots,a_n;\tilde{\delta},i)\boldsymbol{X}_{(i)}\right)^{-\mathbf{1}} \boldsymbol{X}_{(i)}{}'g(a_1,\dots,a_n;\tilde{\delta},i)Y_{(i)}.\right]\right)^2 - 1\right) \qquad (15)$$

Due to the non-linear nature of the objective function, the first order condition obtained by differentiating (15) with respect to $\tilde{\boldsymbol{\delta}}$ cannot be easily solved to express $\tilde{\boldsymbol{\delta}}$ as a function of the data alone.

The basic task of the calibration stage requires estimation of the sample counterpart to the *correspondence parameter* vector, $\boldsymbol{\delta}$. In order to estimate the empirical *calibration parameter* vector, $\hat{\boldsymbol{\delta}}$**,** PLWLS employs the Matlab constrained sequential quadratic programming optimization algorithm, *fmincon*. The *fmincon* algorithm uses a reflective Newton method employing linear constraints (Coleman and Li 1994). The constraints used during estimation are discussed below. The reflective Newton method is a gradient-based search useful for non-linear optimization. The PLWLS application of *fmincon* employs a numerical Hessian and gradient. The *fmincon* algorithm assumes a continuously differentiable objective function and constraints. I assume that the objective function in the optimization problem is continuously differentiable because variations in arguments adjust weights for a weighted least squares regression and a correspondence equation that are both continuously differentiable for positive weights. To further ensure the regression is continuously differentiable with respect to the weight function, rank tests were performed during specification testing with the OLS function estimated in Chapter 4. Optimization outputs were examined to verify that minimums were found within narrow tolerances without the program reporting errors.

The optimization procedure employs a set of linear constraints during the estimation of the *correspondence parameters* in the *fmincon* Matlab routine. The constraints are expressed in equation (16).

$$\begin{pmatrix} 0 \\ 0 \\ \vdots \\ 0 \end{pmatrix} \leq \tilde{\boldsymbol{\delta}} \leq \begin{pmatrix} 100 \\ 100 \\ \vdots \\ 100 \end{pmatrix} \tag{16}$$

Due to the linear restrictions defined in (16), parameter estimates will be neither non-negative nor large and correspondence weights will be between 0 and 1. The non-negativity restriction is implemented based on the theoretical notion that an increase in the distance between the *correspondence attributes* is expected to not provide additional information about the centered observation. The restriction that estimated *correspondence parameters* are not large is implemented to avoid searching for extremely large parameter values, which is useful for ensuring convergence.

Once *correspondence parameters* have been estimated, I use equation (17), a variant of equation (13), to calculate a forecast weight matrix, $\widetilde{\mathbf{W}}_{\sim\mathrm{i}}$, using an out-of-sample vector of correspondence attributes, $\boldsymbol{a}_{h\sim i}$, for an out-of-sample observation, denoted with the index $\sim i$. At this point in the process, data about the unstudied policy site, such as the case study NWRs, is used to construct correspondence weights.

$$g\left(\boldsymbol{a}_{\sim i}, \boldsymbol{a}_1, \ldots, \boldsymbol{a}_n; \widetilde{\boldsymbol{\delta}}, \sim i\right) =$$

$$diag\left\{exp\left(-\begin{bmatrix} |\mathrm{a}_{1\sim i} - \mathrm{a}_{11}| & |\mathrm{a}_{2\sim i} - \mathrm{a}_{21}| & \ldots & |\mathrm{a}_{H\sim i} - \mathrm{a}_{H1}| \\ |\mathrm{a}_{1\sim i} - \mathrm{a}_{12}| & |\mathrm{a}_{2\sim i} - \mathrm{a}_{22}| & \ldots & |\mathrm{a}_{H\sim i} - \mathrm{a}_{H2}| \\ \vdots & \vdots & \ddots & \vdots \\ |\mathrm{a}_{1\sim i} - \mathrm{a}_{1n}| & |\mathrm{a}_{2\sim i} - \mathrm{a}_{2n}| & \ldots & |\mathrm{a}_{H\sim i} - \mathrm{a}_{Hn}| \end{bmatrix}\begin{bmatrix} \tilde{\delta}_1 \\ \tilde{\delta}_2 \\ \vdots \\ \tilde{\delta}_H \end{bmatrix}\right)\right\} \qquad (17)$$

$$= \widetilde{\mathbf{W}}_{\sim \mathrm{i}}$$

The out-of-sample estimated PLWLS weight matrix, $\widetilde{\mathbf{W}}_{\sim \mathrm{i}}$, is then used in a conventional WLS equation with the original full data matrix $\boldsymbol{X}$ and vector Y to estimate a PLWLS parameter specific for forecasting the dependent variable of the out-of-sample observation. The WLS estimator and dependent variable forecast with the full data are as follows,

$$\begin{aligned} \tilde{B}_{\sim i} &= (\boldsymbol{X}'\widetilde{\mathbf{W}}_{\sim i}\boldsymbol{X})^{-1}\boldsymbol{X}'\widetilde{\mathbf{W}}_{\sim i}Y \\ \mathrm{y}_{\sim \mathrm{i}} &= \mathrm{X}_{\sim \mathrm{i}}\tilde{B}_{\sim i} = \mathrm{X}_{\sim \mathrm{i}}(\boldsymbol{X}'\widetilde{\mathbf{W}}_{\sim i}\boldsymbol{X})^{-1}\boldsymbol{X}'\widetilde{\mathbf{W}}_{\sim i}Y. \end{aligned} \qquad (18)$$

Here $\mathrm{y}_{\sim \mathrm{i}}$ is the scalar, out-of-sample dependent variable forecast and $\mathrm{X}_{\sim \mathrm{i}}$ is the $1 \times k$ row vector of independent variable values for the out-of-sample observation. In the next section, I describe the method for estimating the H element vector of correspondence parameters, $\widetilde{\boldsymbol{\delta}} = \left[\tilde{\delta}_1, \ldots, \tilde{\delta}_H\right]'$ and producing out-of-sample forecasts as a sequence of steps, divided into a correspondence stage and a forecast stage.

A Step-by-Step Exposition of the Calibration Stage

The optimization that occurs in the calibration process can be understood as an iterative process that contains several steps that occur with each iteration. The flow chart in Figure 14 depicts the iterative calibration stage in the top box and the non-iterative forecast stage in the bottom box. The optimization process of the calibration stage begins at C1 with initial *correspondence parameter* values that result in conventional, evenly weighted OLS. This is accomplished by first setting all elements of the correspondence parameter vector $\widetilde{\boldsymbol{\delta}}$ to zero; by doing so the process begins with values that ensure that the PLWLS estimator can do no worse than single equation OLS for modeling local residuals. Importantly, with the objective function in equation (12) there is no guarantee of reduced transfer error.

In step C2 the *correspondence attributes* and the conjectured value of $\tilde{\boldsymbol{\delta}}$ are used to construct the regression weights. These weights are calculated with the scalar *correspondence weight equation* as shown in equation (19) below.

$$\tilde{\mathrm{w}}_{\mathrm{ij}} = \tilde{\mathbf{W}}_{\mathrm{ijj}} = exp^{-\sum_{h=1}^{H}|a_{hi}-a_{hj}|\tilde{\delta}_h} = exp^{-(|a_{1i}-a_{1j}|\delta_1+\cdots+|a_{Hi}-a_{Hj}|\tilde{\delta}_H)} \tag{19}$$

Equation (19) is the scalar version of equation (13) above. Here w_{ij} or equivalently $\tilde{\boldsymbol{W}}_{\mathrm{ijj}}$ is the diagonal element of the $\big((n-1)\times(n-1)\big)$ weight matrix, $\tilde{\boldsymbol{W}}_i$, applied to observation j in the WLS regression centered on observation i. Because I omit the centered observation from each centered regression, this step results in $n(n-1)$ weights, that is there are $n-1$ weights for each of n centered regression equations. Additionally, because the *correspondence weight function* takes the absolute value of differences, each weight is repeated once (i.e., $|a_{hi}-a_{hj}|\delta_h = |a_{hj}-a_{hi}|\,\delta_h$) resulting in $n(n-1)/2$ potentially unique regression weights.

Step C3 of each iteration of the calibration stage uses the *correspondence weights* in equation (19), which are based on the conjectured values for the sample *correspondence parameters*, to estimate n weighted regressions and n^2-n local residuals. These local residuals are found by first using a WLS regression to estimate a parameter vector, $\tilde{B}_i$, for each of the n centered regression equations,

$$\tilde{B}_i = (\boldsymbol{X}_{(i)}{}'\tilde{\mathbf{W}}_i\boldsymbol{X}_{(i)})^{-1}\boldsymbol{X}_{(i)}{}'\tilde{\mathbf{W}}_iY_{(i)}. \tag{20}$$

Next in step C3, the n estimated models from (20) are employed to calculate local residuals for the $n-1$ temporarily un-centered observations. The n^2-n local residuals, $\tilde{e}_{ij}$, can be expressed as follows,

$$\tilde{e}_{ij} = Y_j - X_j\tilde{B}_i. \tag{21}$$

As mentioned previously, the (i) subscript in equation (20) simply indicates that the row of data for the centered or i^{th} observation are dropped. The dropping of data occurs only during the calibration stage of the PLWLS estimator because the *correspondence weight equation* in (13) and (19) is not intended for predicting the correspondence between an observation and itself.

In step C4, the weights and residuals are used in the estimated *correspondence equation* from (11) to calculate the $(n-1)$ values of the correspondence residuals, $\tilde{u}_{ij}$, for each of the n centered equations. These *correspondence residuals* provide $n^2 - n$ estimates of u_{ij} for the objective function to be paired with the same number of estimated *correspondence weights*, $\tilde{w}_{ij}$, in the objective function from equation (12).

Step C5 is where the *fmincon* algorithm evaluates the conjectured *correspondence parameter* vector, $\tilde{\boldsymbol{\delta}}$, to determine if the objective function is at a minimum. If the minimization criteria are not met, the algorithm determines the next guess of the *correspondence parameter*s. If the *fmincon* algorithm determines that a minimum has been found, $\tilde{\boldsymbol{\delta}}$ is retained as the PLWLS *correspondence parameter* vector and the forecasting stage can then be used for benefit transfers for policy sites that may lack primary value estimates.

A Step-by-Step Exposition of the Forecast Stage

The entire optimization procedure described in the preceding section was used entirely for estimating the H-element *correspondence parameter* vector, $\tilde{\boldsymbol{\delta}}$. The H scalar *correspondence parameter* values are global and thus apply to all n centered regression models and to any forecast models. After estimating the vector of *correspondence parameters* in the calibration stage, the next step required for an out of sample forecast is estimation of forecast regression parameters which require the estimated *correspondence parameters*, the in-sample data, and the out-of-sample correspondence attributes associated with the benefit transfer forecast of interest. The procedure is more complicated than OLS or WLS forecasting, but the same basic idea is present: first, estimate a regression using in-sample data and second, use the estimated regression with out-of-sample variables to forecast the dependent variable. The difference that arises when using the PLWLS model for forecasting is that out-of-sample *correspondence attributes* are required to supplement the in-sample data, as indicated in equation (17), for estimating the correspondence weights that are used during the WLS estimation step from equation (18).

The process for generating a regression for an out-of-sample observation is much like the calibration stage. In step F1, data for the policy site is obtained, including the out-of-sample, k-element row vector of independent variable values, $X_{\sim i}$, and the out-of-sample column vector of H *correspondence attributes*, $\boldsymbol{a}_{\sim i}$. The data in the in-sample correspondence attribute vectors, $\boldsymbol{a}_1, \ldots, \boldsymbol{a}_n$ and the out-of-sample correspondence attribute vector $\boldsymbol{a}_{\sim i}$ are used to calculate the out-of-sample *correspondence attribute* weight matrix, $\widetilde{\mathbf{W}}_{\sim i}$, composed of n scalar weights, $\widetilde{w}_{\sim ij}$, for forecasting the centered, out-of-sample regression and dependent variable as indicated in equation (18).

In step F2, the post-calibration, forecasting version of the scalar correspondence weight equation found in equation (22) demonstrates how each individual weight is calculated using the out of sample correspondence attributes.

$$\begin{aligned} \widetilde{\mathrm{w}}_{\sim \mathrm{ij}} = \widetilde{\mathbf{W}}_{\sim i\, jj} = e^{-\sum_{h=1}^{H} |a_{h\sim i} - a_{hj}| \tilde{\delta}_h} \\ = e^{-(|a_{1\sim i} - a_{1j}| \tilde{\delta}_1 + \cdots + |a_{H\sim i} - a_{Hj}| \tilde{\delta}_H)} \end{aligned} \tag{22}$$

In equation (22), the jj subscript on $\widetilde{\mathbf{W}}_{\sim i}$ indicates the j^{th} diagonal element of the $n \times n$ forecast weight matrix.

In step F3, the $(n \times n)$, diagonal forecast *correspondence weight* matrix, $\widetilde{\mathbf{W}}_{\sim i}$, (or the n scalar values, $\widetilde{w}_{\sim ij}$) from F2 and the full data matrix $\boldsymbol{X}$ and vector Y are used to construct the PLWLS estimator for the out-of-sample observation denoted by $\sim i$ according to equation (23).

$$\tilde{B}_{\sim i} = (\boldsymbol{X}'\widetilde{\mathbf{W}}_{\sim i}\boldsymbol{X})^{-1}\boldsymbol{X}'\widetilde{\mathbf{W}}_{\sim i}Y \tag{23}$$

Finally, in step F4, the forecast for the out-of-sample observation is computed just as with WLS according to the expression in equation (24).

$$\begin{aligned} \tilde{y}_{\sim i} = X_{\sim i}\tilde{B}_{\sim i} \\ = X_{\sim i}(\boldsymbol{X}'\widetilde{\mathbf{W}}_{\sim i}\boldsymbol{X})^{-1}\boldsymbol{X}'\widetilde{\mathbf{W}}_{\sim i}Y \end{aligned} \tag{24}$$

In contrast to the PLWLS calibration stage discussed above, in equation (24) the full, in-sample data matrix $\boldsymbol{X}$ and vector Y are used to calculate the final out-of-sample forecast regression parameter, $\tilde{B}_{\sim i}$.

Marginal Effects

Marginal effects are the partial derivatives that indicate the change in the dependent variable that is due to the change in any of the explanatory or independent variables, typically estimated holding all other variables constant. Calculation of marginal effects for the PLWLS model is complicated, relative to the OLS model, by the estimation of *correspondence parameters*. If interest lies in estimating marginal effects of a variable for a particular observation one can simplify the calculation by assuming that *correspondence weights* are fixed, which implies that both the *correspondence attributes* (i.e., $\boldsymbol{a}_i$) and the calibrated *correspondence parameters* (i.e., $\tilde{\delta}_1, \ldots, \tilde{\delta}_H$) are fixed. Alternatively, one can hold the regression parameters fixed and allow *correspondence parameters* to vary; and both conditions can be relaxed. Ultimately, because *correspondence attributes* (some of which are also explanatory variables), *correspondence parameters*, explanatory variables, and regression parameters can all vary there are a number of definitions of marginal effects available.

I estimate the marginal effect on the dependent variable of changes in explanatory variables for two sources of variation, changes in *correspondence attributes* and changes in explanatory variables. Both types of marginal effects are with respect to explanatory variables, but due to the nature of the PLWLS model, the marginal effect of a variable may change across both explanatory variable and *correspondence attribute* space. For the reported results when I allow explanatory variables to change, *correspondence attributes* are held constant and vice versa.

In the results, I present graphs of both types of marginal effects using the 82 sets of estimated *correspondence parameters* from the jackknife simulation. Doing so introduces additional variability such that patterns identified in the marginal effects are more robust to resampling variability than if I were to use the single set of full data *correspondence parameters*.

The examination of the marginal effects of changes in explanatory variables across the explanatory variable space focuses on the estimated, in-sample marginal effects or parameter estimates obtained from the jackknife simulation. The jackknife analysis provides 82 observations of marginal effects for each variable that can be displayed in a variety of sequences to highlight patterns.

The examination of the marginal effects of explanatory variables that may shift with *correspondence attributes* focuses on out-of-sample forecasts for the 4 case study NWRs. These values may be unique with respect to the in-sample estimates due to differing attribute levels at the case study sites. In the event that a case study site had the same *correspondence attribute* values as an in-sample observation, the centered regression equations for two observations would be identical.

Results

As the calibration of *correspondence parameters* is the first step in the PLWLS estimator, I present these values first. The results of the PLWLS calibration can be found in Table 16. All *correspondence attributes* were standardized by their sample means and variances prior to calibration, with the exception of Euclidean distance which was converted from meters to 1000's of kilometers. In the table the standardization of *correspondence attributes* other than distance has been reversed to facilitate a comparison. The results of the calibration stage indicate that GDP and Euclidean distance between sites are important non-method-related determinants of correspondence among studies in the sample. Additionally, studies that value similar services and which aggregate over a similar population have higher correspondence with each other than otherwise. The PLWLS calibration stage found that the number of acres valued by a study was not a determinant of correspondence, a notable result that might be reassessed in future studies. The moderate values for these parameters and the moderate weights (typically between 0.01 and 0 .9) that are consequently applied in each regression suggest that the algorithm is moderately down-weighting observations with poor correspondence, but retaining ample information for reasonably robust estimation.

The continuous *correspondence parameters* can be interpreted somewhat like elasticities. For the GDP per capita variable, for example, the estimated *correspondence parameter* value of 1.28 can be interpreted as follows: a 1% increase in the difference in GDP per capita between a study site and a policy site leads to about a 1.28% decrease in the weight applied to the study site in the regression model for the policy site. The binary variables have similar interpretations. For example, the flood control/storm protection parameter indicates that when forecasting a flood control/storm protection study at a policy site, a study site valuing the same service has a higher weight than study sites that value different services.

The next step in the PLWLS procedure after identifying *correspondence parameters* is to use those parameters to calculate a $(n \times n)$ matrix of regression weights for each out-of-sample study/site of interest. These regression weights are used to calculate a regression for each site and also can be used to rank observations according to their relative information content. For any given centered site, observations that are weighted more heavily are assumed to have greater correspondence and therefore be more informative about the centered site.

Several graphs illustrate correspondence in 2 dimensions; these graphs specifically depict the NWR study sites as the centered observation in each panel in Figure 15. Figure 15 contains a plot for each refuge, where each refuge is the centered site indicated by the open circle towards the top-right of each graph, which is also the graph's origin. In each graph the horizontal axis represents the weighted difference between each data point and the centered observation. Filled circles that are colored blue receive lower weights, while magenta circles receive higher weights, indicating greater correspondence. The four smaller filled circles represent the four case study NWRs, which can be seen to all have the same population, but varying incomes. The purpose of the figure is to demonstrate that the *correspondence parameters* give rise to weights that display a similar pattern across these out-of-sample observations or policy sites. The figure also demonstrates how correspondence can be thought of in a multi-dimensional space. Future research is needed to develop an application for this information.

While the OLS regression treats each observation equally, the PLWLS approach allows one to rank observations according to the weight applied in each centered regression. The highest ranked observations contribute the most information, so the forecasted primary valuation study results are associated most strongly with the characteristics of the most highly weighted observations. Tables 17-24 contain the ten observations in each centered regression receiving the highest weights for each of the 4 case study refuges for both water quality and flood control/storm protection services. The first four tables are for water quality at each site and the next four are for flood control/storm protection. The top row of each table contains the values for the centered, policy site NWR followed in the second row by the column headings. GDP per capita values are all in 2010 US dollars. Euclidean distances are in kilometers. Fo r these tables, the regression weights are normalized so that all n=82 observations have weights that sum to 82; this normalization (which is not used during estimation) facilitates a simple comparison with OLS weights that are always 1 by assumption. Some rows in these tables appear to be duplicates, which is a result of variations in site attributes not included in the table such as the number of acres of wetlands valued. For example, in Table 17 the first 3 observations are from Bishop et al. (2000), which differ in both the measured value of WTP/person/acre/year and the number of wetland acres valued, which are both variables not included in these tables because those variables are not modeled as *correspondence attributes*.

Generally, one can see that studies that value the same service are always the highest ranked observations for all 8 examples. High correspondence observations that value the same service as the centered observation tend to have markedly higher weights than subsequent observations that value other services; however this is not always the case. In Table 21, for example, the weight applied to the Costanza, Farber, and Maxwell (1989) flood control/storm protection study is substantially lower than the weights applied to the higher correspondence flood control/storm protection studies, which is consistent with the non-zero *correspondence parameters* estimated for GDP per capita and Euclidean distance. Both of these variables differ considerably more between Arrowwood NWR and Costanza et al.'s (1989) Louisiana study site and the higher ranked prairie pothole observations of Leitch and Hoyde (1996) and Roberts and Leitch (1997), as can be seen in the table.

The valuation forecasts of a stated preference study of flood control and water quality benefits for each of the NWR policy sites can be found in Tables 25 and 26. The former table contains the median and mean of the dependent variable after reversing the log transformation and the values are scaled to 2010 US dollars per thousand acres per thousand people. The latter table contains median and mean aggregate values where the population is specified as the mean value in the data set, about 3.5 million people. The median population is about half as large; the mean population in the data set is relatively high due to several observations that aggregate benefits over large populations. The median WTP values are the point estimates of WTP forecasted by OLS and PLWLS estimators. For each estimator I include a forecast of both the median and mean values of willingness to pay. The magnitude of the difference between the median and the mean is inversely proportional to the precision of each model, and the consistently smaller gap between mean and median for the PLWLS model implies that this model is consistently more precise than the OLS model. Table 26 also provides acreage counts used during estimation and aggregation. Also of note, many of the PLWLS dependent variable estimates, both mean and median, fall between the mean and median of the OLS estimates, implying that the PLWLS results do not suffer from extensive bias relative to the results from the unbiased OLS estimator. I highlight the mean value single service revealed preference study results as I consider these to be the best estimates (i.e., for forecasting the value of ecosystem services when estimated by a primary valuation study) available from the meta-analysis regression method. Under the suggested interpretation of the forecasts of the dependent variable, best estimates include the judgment that a stated preference study of a population with about 3.5 million people would be the best choice to estimate the total economic value of each service.

Jackknife Leave-One-Out Analysis

The results of the jackknife leave-one-out procedure indicate that the additional information used in the PLWLS forecast may be useful for lowering out-of-sample forecast variances. The results indicate that of 82 observations, the PLWLS algorithm when forecasting an out-of-sample observation produces a lower variance estimate than OLS 50 times (61%). The leave-one-out forecasts of the PLWLS estimator beat the full data forecasts of OLS 35 out of 82 times (43%), while the leave-one-out forecasts of the OLS estimator only beat the full information version of the PLWLS estimator 18 out of 82 times (22%). Future work may focus on identifying patterns that could be useful for predicting whether OLS or PLWLS is more likely to produce a more efficient out-of-sample forecast.

In Figure 16 the 8 NWR meta-analysis benefit transfer estimated values from each of 3 estimators are arranged along the horizontal axis. Labels are excluded from these as the primary purpose of the graphic is to demonstrate the relative efficiency of forecasts. Each small dot represents the natural logarithm of WTP for the appropriate service and refuge based on a single iteration of the jackknife procedure. The 3 panels are arranged in order of increasing apparent forecast variability. The first panel contains the PLWLS forecasts, labeled as "gls-style" to emphasize the functional form of the objective function in equation (12). The middle panel contains OLS forecasts, and the far right panel contains forecasts from an alternative functional form of the PLWLS estimator where parameters are chosen to minimize n squared transfer errors in the spirit of OLS. Figure 16 serves to enable a graphical approach to operational validation (Sargent 2013) of the efficiency claims associated with the PLWLS estimator relative to OLS and an alternative specification of PLWLS that minimizes transfer error rather than variance forecast error. One can also see from this figure that PLWLS does not appear to suffer from bias relative to the OLS estimates.

Marginal Effects

As mentioned above, a number of options exist for estimating marginal effects. First, Figure 17 contains jackknife simulated parameters for each centered regression. These parameters are both elasticity estimates and marginal effect estimates because the population variable is in log form. Due to these parameters coming from the jackknife simulation, each data point is a parameter for a centered observation when that observation is treated as being out-of-sample. The population variable indicates the number of beneficiaries over which the original primary valuation study sampled and to whom the original researchers attributed value estimates. As indicated in the top panel of the figure, observations are arranged from small to large benefitting populations. While there are apparent trends in the values, the shape of the curve appears to be non-linear. Moreover, one can see that observations with similar associated population sizes cluster together. This clustering is a pattern I expect to find with a local regression approach.

Figures 17 and 18 have a parallel construction. The top panel in each figure contains jackknife forecasts organized along the horizontal axis by the explanatory variable value listed on the vertical axis. Each jackknife replication contains a WLS regression that estimates a parameter vector for the omitted observation. The top panel characterizes the omitted observation while the bottom panel characterizes the parameter estimate obtained in the WLS regression for the omitted observation. The bottom panel contains the same ordering of jackknife replications as the top panel, but with a different value on the vertical axis. The idea behind this setup is that one can visually inspect the figures to look for evidence of correspondence.

Figure 17 contains jackknife replications used to illustrate the variation in the estimated regression parameter for user population. The top panel contains 82 estimates of a population parameter, one for each iteration of the jackknife and forecasted for the omitted observation, arranged in order of increasing user population size. The bottom panel contains the same ordering, but presents the parameter estimates rather than the explanatory variable value. The clustering of parameter estimates for neighboring observations in the bottom panel demonstrates the presence of correspondence.

Figure 18 contains jackknife replications where the vertical bars in the top panel show whether the omitted observation was an estimate of total value. In the top panel, if the omitted study was an estimate of total value of a wetland ecosystem, the bar is tall, so the first 41 observations are not total value studies and the next 41 observations are total value studies. The figure indicates that the regressions for observations that are total value studies more precisely estimate the total value parameter associated with the discrete valued total value dummy variable. While the estimated PLWLS model does not contain a correspondence parameter for total value, because these studies comprise much of the data, studies that value water quality enhancements and flood control/storm protection (services for which a *correspondence parameter* are estimated) are downweighted in the WLS models generated by PLWLS for total value forecasts.

The second type of marginal effect explored in this chapter concerns the effect of a change in a *correspondence attribute*, holding all explanatory variables and *correspondence parameters* constant. Figure 19 contains simulated marginal effects for a sequence of regressions centered on a hypothetical primary valuation study of water quality enhancements provided by wetland at Arrowwood NWR. For each of the three lines, only the value of the *correspondence attribute* indicated by the legend varies; the explanatory variables are held constant. The variation in each *correspondence attribute* is over the range of values in the sample and the range is divided into 1,000 steps to generate a visually continuous line.

All three lines plotted in Figure 19 have values on the vertical axis that can be interpreted as elasticities. The highly elastic and negative value estimated for the impact of income in the state comes as a surprise, as this suggests that water quality enhancements are an inferior good. The remaining elasticities are more consistent with my expectations, as the dependent variable is willingness to pay normalized by population and wetland area. These results indicate that the total value of wetlands increase at a decreasing rate as acreage increases or as the population for which benefits are estimated increases.

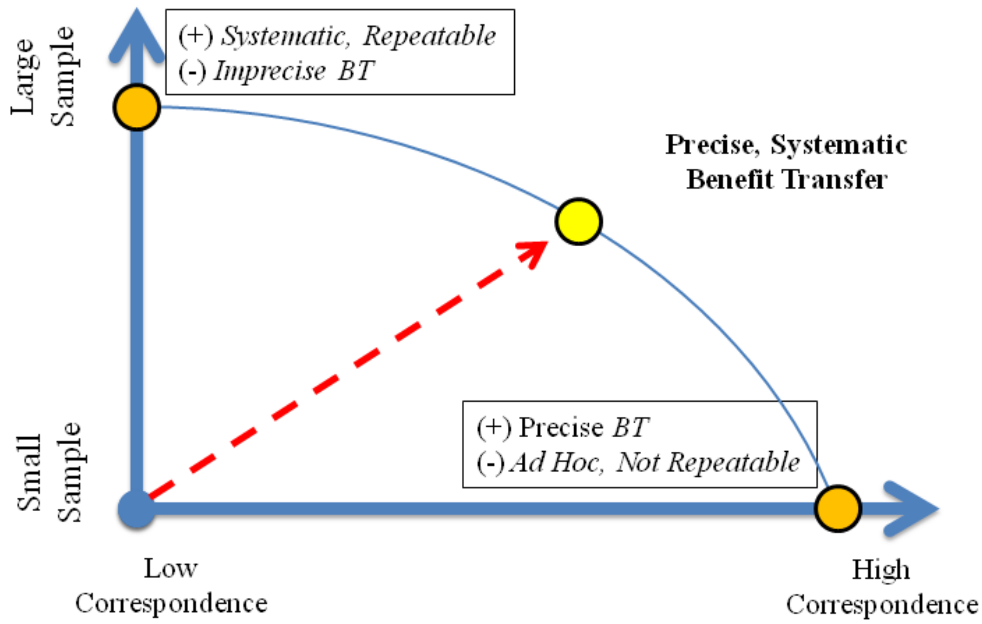


Figure 13. A Conceptual diagram of correspondence-sample size frontier, illustrating the trade-offs between correspondence and sample size in achieving a precise and systematic model for meta-analysis benefit transfer

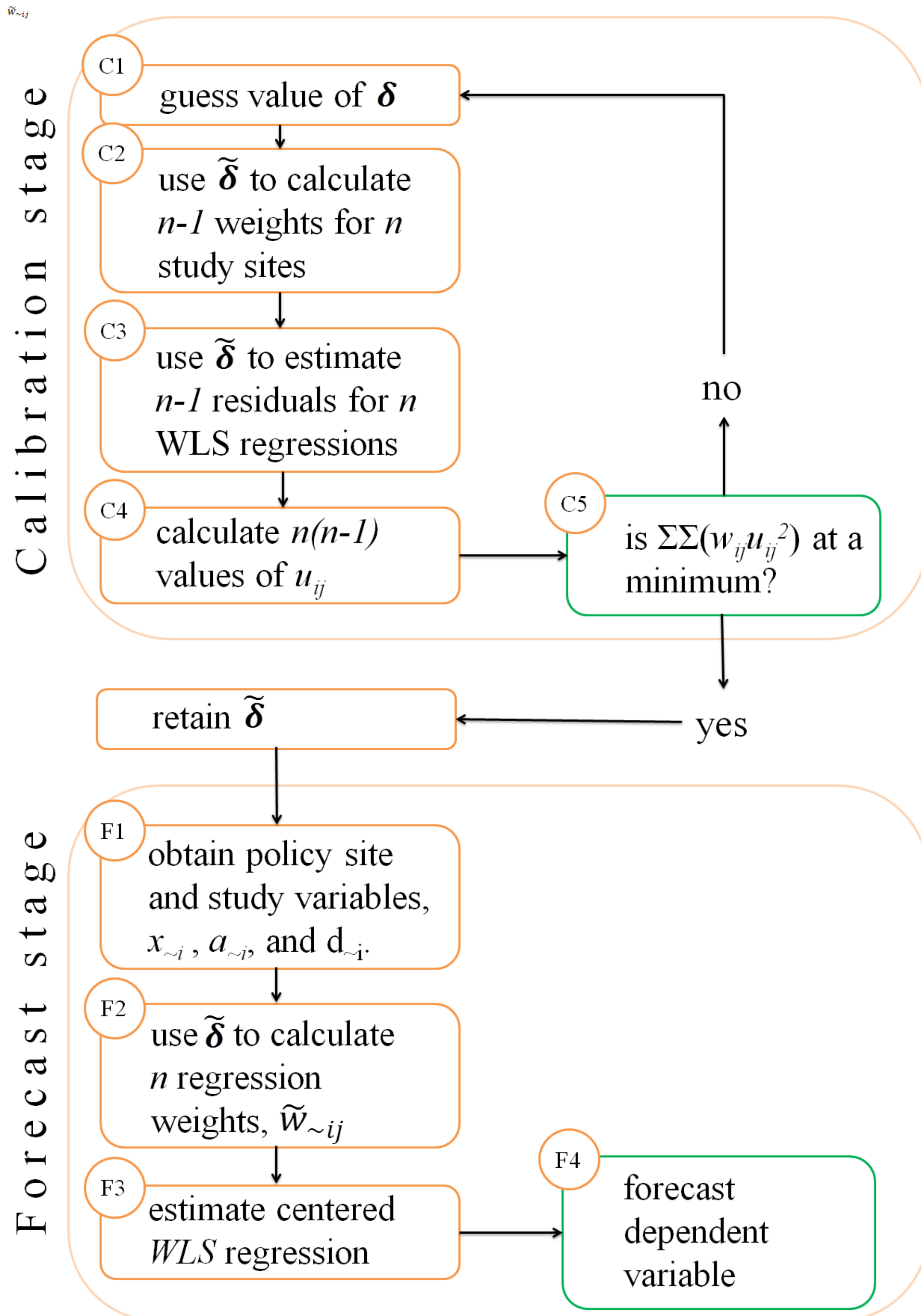


Figure 14: Steps in the PLWLS modeling process

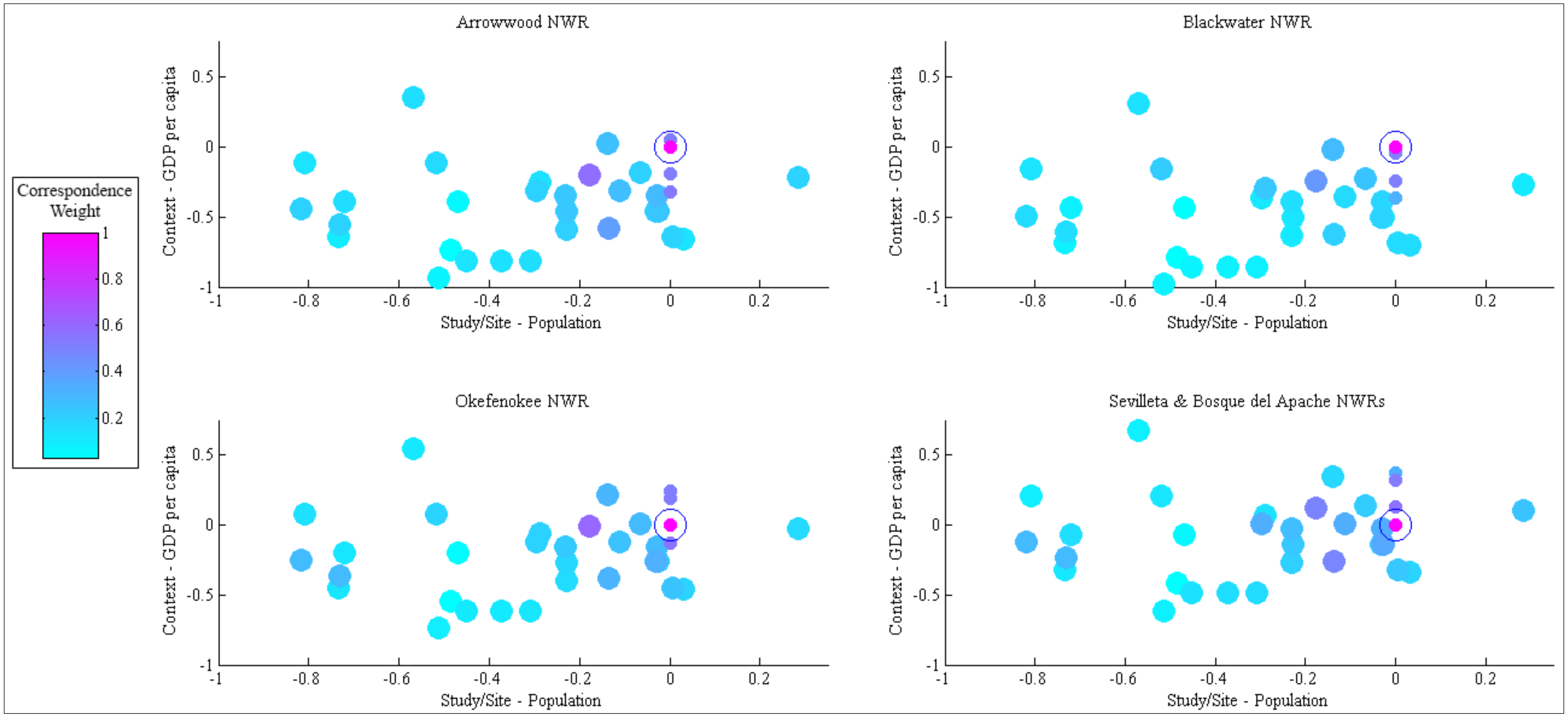
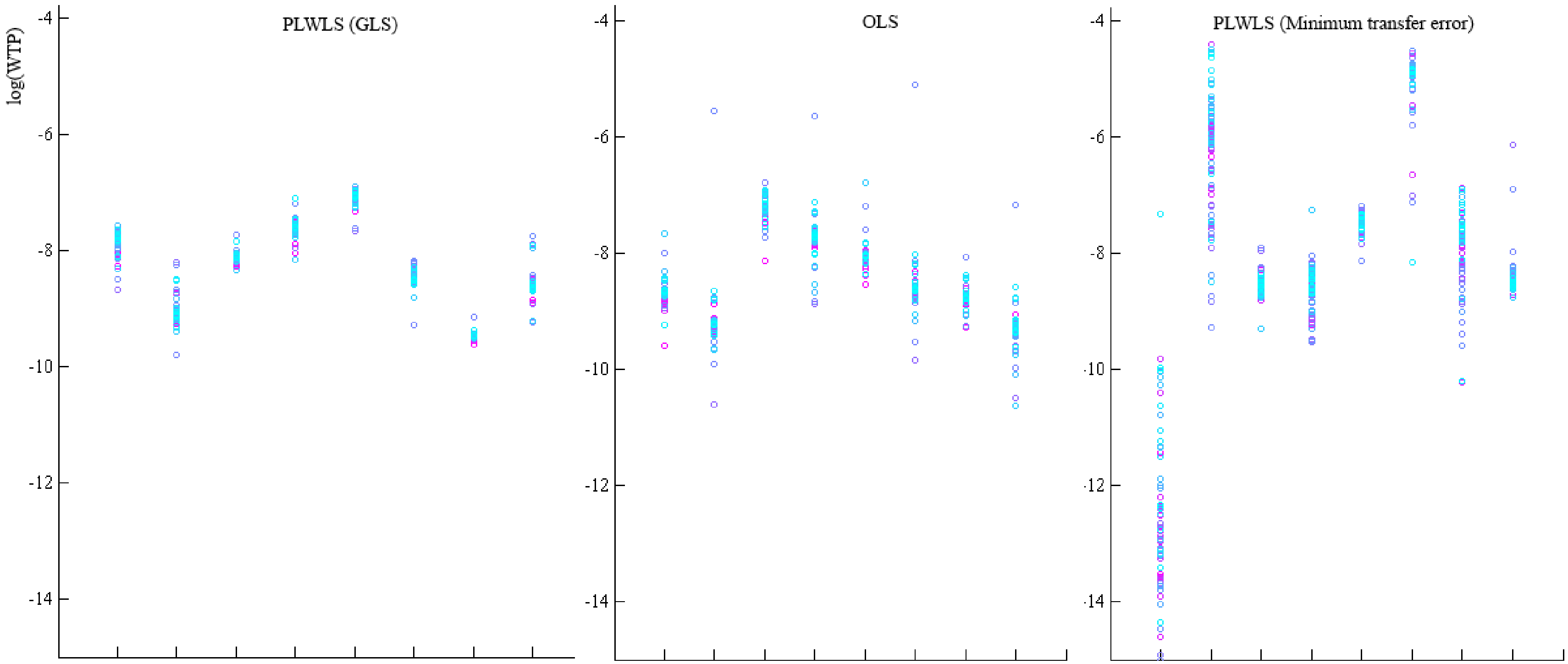
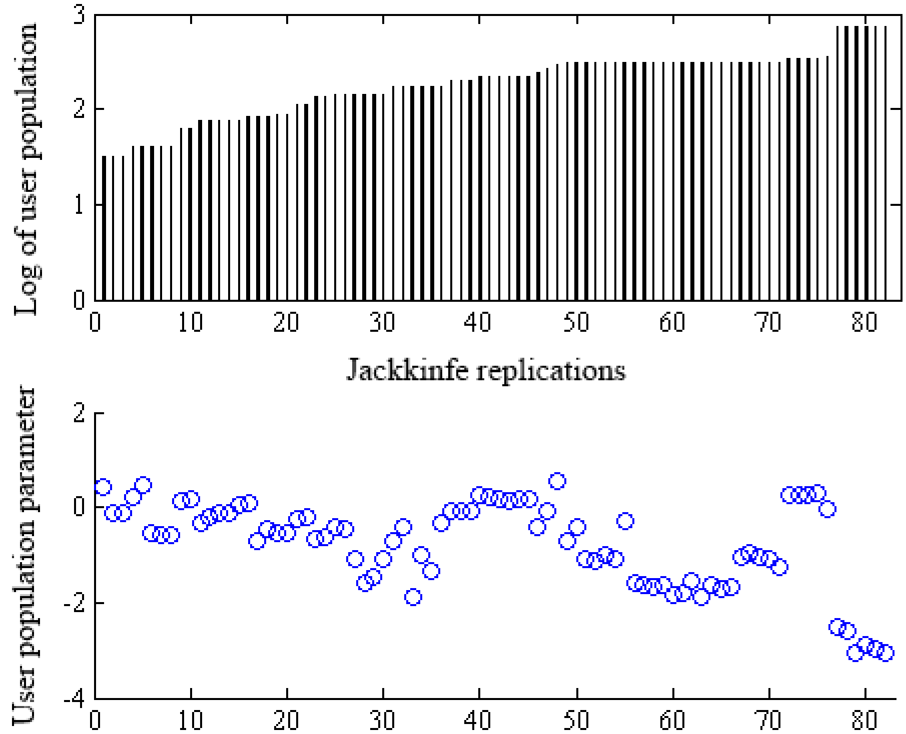


Figure 15: Plots of *Correspondence Distance* for 4 Case Study NWRs for stated preference water quality valuation



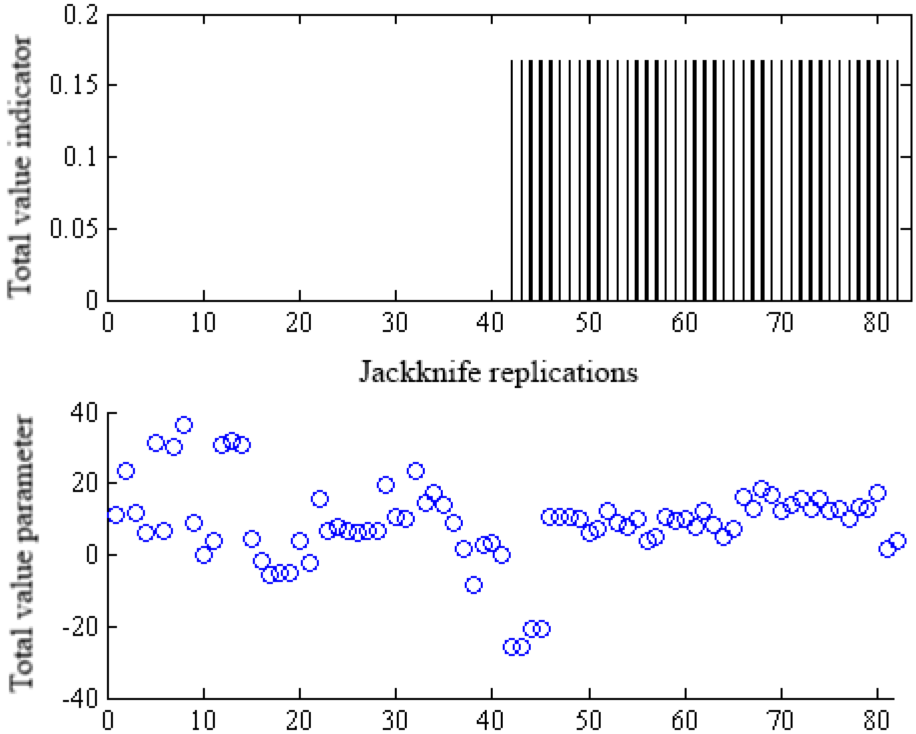
This figure demonstrates the variability of three estimators. The figure contains benefit transfer forecasts for three methods for each service and each case study NWR. Vertical axes represent the log of forecasted WTP and the horizontal axis contains jackknife replications of the forecasted value where each pair is for a single case study NWR in alphabetical order and each item of each pair is a forecast of water quality enhancement value and flood control/storm protection, respectively.

Figure 16: Jackknife graphical validation of PLWLS forecast efficiency



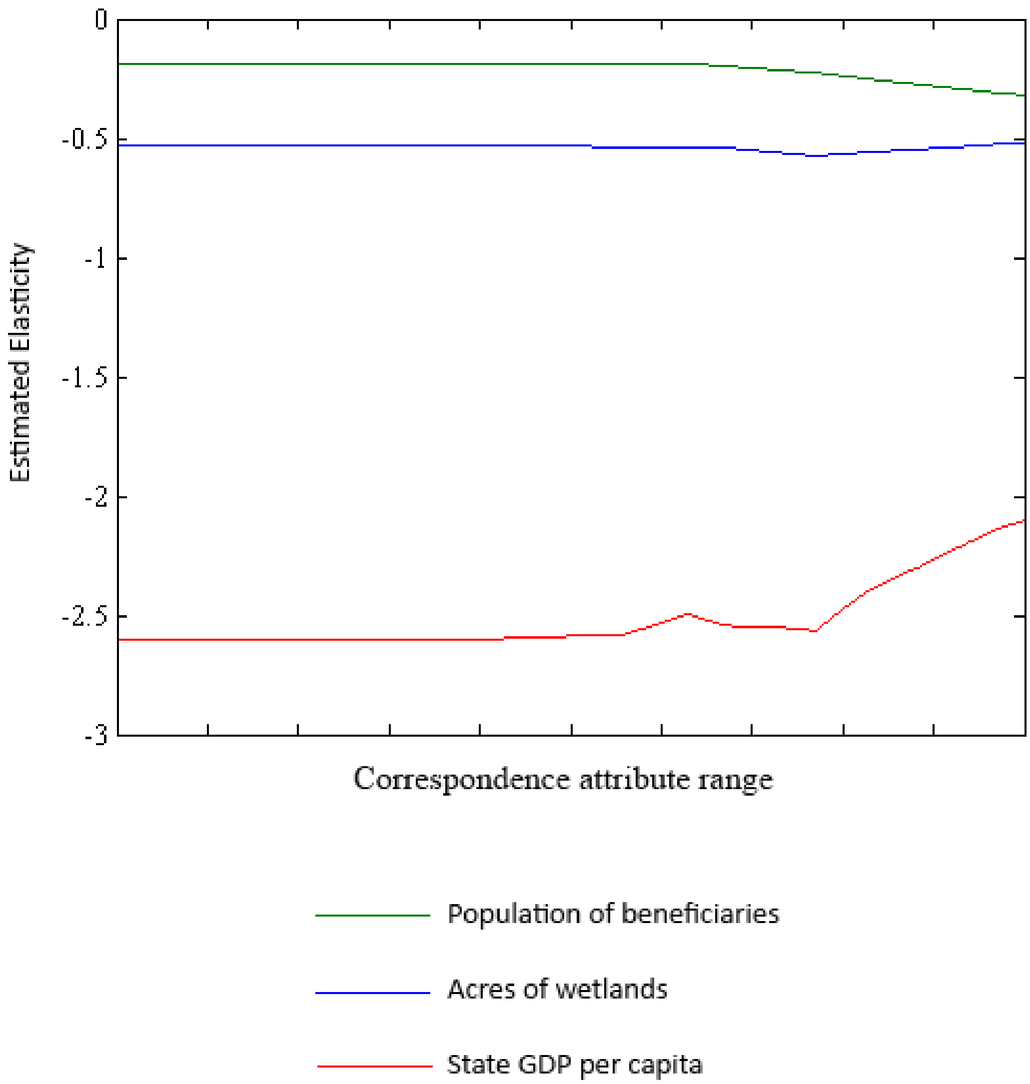
Top panel contains user population value for omitted observation for jackknife replications ordered in increasing value. Bottom panel contains the same ordering of omitted observations with the PLWLS parameter estimate for user population for the omitted observation. The figure is intended to illustrate correspondence among studies with similar user populations.

Figure 17: Jackknife simulation of local regression parameter for the population of beneficiaries



Top panel contains total value dummy variable for omitted observation for jackknife replications ordered in increasing value. Bottom panel contains the same ordering of omitted observations with the PLWLS parameter estimate for the total value dummy variable for the omitted observation. The figure is intended to illustrate correspondence among studies that value the same service.

Figure 18: Jackknife simulation of local regression parameter for the dummy variable indicating observations that estimate the total value of a wetland ecosystem



This figure illustrates how parameter estimates vary as each of three correspondence attributes varies. Each line is generated by varying in small increments the associated correspondence attribute across the range of attributes present in the sample and forecasting a WLS regression for a water quality valuation study at Arrowwood NWR using that attribute value. *Correspondence parameters*, explanatory variables, and all other *correspondence attributes* are held constant.

Figure 19: Simulated variation in marginal effects due to change in *correspondence attributes* for the regression model centered on a stated preference water quality valuation study at Arrowwood NWR

Table 16: PLWLS Estimated *Correspondence parameters*

Correspondence attribute	**Estimated *Correspondence parameter***
Flood control/storm protection	1.365992
GDP	1.276706
Water quality	0.760224
Coastal	0.218638
Distance 10^3km	0.194442
Population	0.133756
Acres	0

Table 17: Arrowwood NWR Stated Preference Water Quality, Ten Highest Correspondence Observations

The top row of the table contains the policy site. Lower rows contain the ten observations with the highest predicted *correspondence weights*. Repeated studies (e.g., Bishop et al. 2000) represent unique value estimates that may share *correspondence attributes*.

Arrowwood NWR	Stated Pref.	Water quality	$42,500	0 km	3,531,251	N/A
Author-date	**Method**	**Service valued**	**GDP per capita**	**Euclidean distance**	**Study pop.**	**Regression Weight**
Bishop et al. 2000	Stated pref.	Water quality	$36,200	917	936,090	2.7697
Bishop et al. 2000	Stated pref.	Water quality	$36,200	917	936,090	2.7697
Bishop et al. 2000	Stated pref.	Water quality	$36,200	917	936,090	2.7697
Sutherland and Walsh 1985	Stated pref.	Water quality, use value	$27,000	1162	1,275,514	1.8876
Sutherland and Walsh 1985	Stated pref.	Water quality bequest value	$27,000	1162	1,275,514	1.8876
Sutherland and Walsh 1985	Stated pref.	Water quality exist. value	$27,000	1162	1,275,514	1.8876
Sutherland and Walsh 1985	Stated pref.	Water quality option value	$27,000	1162	1,275,514	1.8876
Sutherland and Walsh 1985	Stated pref.	Water quality total value	$27,000	1162	1,275,514	1.8876
Phaneuf and Herriges 1999	Revealed pref.	Recreation	$32,300	645	630,561	1.3689
Phaneuf and Herriges 1999	Revealed pref.	Recreation	$32,300	721	630,561	1.3488

Table 18: Blackwater NWR Stated Preference Water Quality, Ten Highest Correspondence Observations

The top row of the table contains the policy site. Lower rows contain the ten observations with the highest predicted *correspondence weights*. Repeated studies (e.g., Bishop et al. 2000) represent unique value estimates that may share *correspondence attributes*.

Blackwater NWR	Stated Pref.	Water quality	$44,000	0 km	3,531,251	N/A
Author-date	**Method**	**Service valued**	**GDP per capita**	**Euclidean distance**	**Study pop.**	**Regression Weight**
Bishop et al. 2000	Stated pref.	Water quality	$36,200	1180	936,090	2.8578
Bishop et al. 2000	Stated pref.	Water quality	$36,200	1180	936,090	2.8578
Bishop et al. 2000	Stated pref.	Water quality	$36,200	1180	936,090	2.8578
Beran 1995	Stated pref.	Total value	$43,300	695	1,258,753	1.9196
Petrolia and Kim 2011	Stated pref.	Total value	$36,800	1702	2,146,273	1.7119
Bauer, Cyr, and Swallow 2004	Stated pref.	Total value	$34,800	512	408,000	1.6123
Bauer, Cyr, and Swallow 2004	Stated pref.	Total value	$34,800	512	408,000	1.6123
Bauer, Cyr, and Swallow 2004	Stated pref.	Total value	$34,800	512	408,000	1.6123
Bauer, Cyr, and Swallow 2004	Stated pref.	Total value	$34,800	512	408,000	1.6123
Johnston et al. 2002	Stated pref.	Total value	$38,800	433	73,423	1.4913

Table 19: Okefenokee NWR Stated Preference Water Quality, Ten Highest Correspondence Observations

The top row of the table contains the policy site. Lower rows contain the ten observations with the highest predicted *correspondence weights*. Repeated studies (e.g., Bishop et al. 2000) represent unique value estimates that may share *correspondence attributes*.

Okefenokee NWR	Stated Pref.	Water quality	$36,400	0 km	3,531,251	N/A
Author-date	**Method**	**Service valued**	**GDP per capita**	**Euclidean distance**	**Study pop.**	**Regression Weight**
Bishop et al. 2000	Stated pref.	Water quality	$36,200	1632	936,090	2.8787
Bishop et al. 2000	Stated pref.	Water quality	$36,200	1632	936,090	2.8787
Bishop et al. 2000	Stated pref.	Water quality	$36,200	1632	936,090	2.8787
MacDonald, Bergstrom, and Houston 1998	Stated pref.	Habitat	$29,700	319	2,793,672	1.5584
Sutherland and Walsh 1985	Stated pref.	Water quality, use value	$27,000	3298	1,275,514	1.4882
Sutherland and Walsh 1985	Stated pref.	Water quality bequest value	$27,000	3298	1,275,514	1.4882
Sutherland and Walsh 1985	Stated pref.	Water quality exist. value	$27,000	3298	1,275,514	1.4882
Sutherland and Walsh 1985	Stated pref.	Water quality option value	$27,000	3298	1,275,514	1.4882
Sutherland and Walsh 1985	Stated pref.	Water quality total value	$27,000	3298	1,275,514	1.4882
Beran 1995	Stated pref.	Total value	$43,300	326	1,258,753	1.4581

Table 20: Sevilleta and Bosque del Apache NWRs Stated Preference Water Quality, Ten Highest Correspondence Observations

The top row of the table contains the policy site. Lower rows contain the ten observations with the highest predicted *correspondence weights*. Repeated studies (e.g., Bishop et al. 2000) represent unique value estimates that may share *correspondence attributes*.

Sevilleta and Bosque del Apache NWRs	Stated Pref.	Water quality	$32,900	0 km	3,531,251	N/A
Author-date	**Method**	**Service valued**	**GDP per capita**	**Euclidean distance**	**Study pop.**	**Regression Weight**
Bishop et al. 2000	Stated pref.	Water quality	$36,200	2040	936,090	1.936
Bishop et al. 2000	Stated pref.	Water quality	$36,200	2040	936,090	1.936
Bishop et al. 2000	Stated pref.	Water quality	$36,200	2040	936,090	1.936
Sutherland and Walsh 1985	Stated pref.	Water quality, use value	$27,000	1662	1,275,514	1.8982
Sutherland and Walsh 1985	Stated pref.	Water quality bequest value	$27,000	1662	1,275,514	1.8982
Sutherland and Walsh 1985	Stated pref.	Water quality exist. value	$27,000	1662	1,275,514	1.8982
Sutherland and Walsh 1985	Stated pref.	Water quality option value	$27,000	1662	1,275,514	1.8982
Sutherland and Walsh 1985	Stated pref.	Water quality total value	$27,000	1662	1,275,514	1.8982
Sanders, Walsh, and McKean 1991	Revealed pref.	Recreation	$29,600	566	2,889,964	1.3836
Sanders, Walsh, and Loomis 1990	Stated pref.	Total use value	$29,600	566	2,889,964	1.3836

Table 21: Arrowwood NWR Stated Preference Flood control, Ten Highest Correspondence Observations

The top row of the table contains the policy site. Lower rows contain the ten observations with the highest predicted *correspondence weights*. Repeated studies (e.g., Leitch and Hoyde 1996) represent unique value estimates that may share *correspondence attributes*.

Arrowwood NWR	Stated Pref.	Flood control	$42,500	0 km	3,531,251	N/A
Author-date	**Method**	**Service valued**	**GDP per capita**	**Euclidean distance**	**Study pop.**	**Regression Weight**
Leitch and Hoyde 1996	Damage avoidance	Flood control	$26,700	98	638,800	3.8413
Leitch and Hoyde 1996	Damage avoidance	Flood control and habitat	$26,700	145	638,800	3.8062
Roberts and Leitch 1997	Damage avoidance	Flood control and habitat	$31,300	233	106,406	3.6005
Roberts and Leitch 1997	Damage avoidance	Flood control	$31,300	235	106,406	3.5993
Costanza, Farber, and Maxwell 1989	Damage avoidance	Flood control	$23,800	2116	94,393	1.3931
Phaneuf and Herriges 1999	Revealed pref.	Recreation	$32,300	645	630,561	1.371
Phaneuf and Herriges 1999	Revealed pref.	Recreation	$32,300	721	630,561	1.3509
Poor 1999	Stated pref.	Total value	$33,200	739	1,541,253	1.2821
Beran 1995	Stated pref.	Total value	$43,300	2205	1,258,753	1.2542
Sanders, Walsh, and Loomis 1990	Stated pref.	Total use value	$29,600	1034	2,889,964	1.1417

Table 22: Blackwater NWR Stated Preference Flood control, Ten Highest Correspondence Observations

The top row of the table contains the policy site. Lower rows contain the ten observations with the highest predicted *correspondence weights*. Repeated studies (e.g., Roberts and Leitch 1997) represent unique value estimates that may share *correspondence attributes*.

Blackwater NWR	Stated Pref.	Flood control	$44,000	0 km	3,531,251	N/A
Author-date	**Method**	**Service valued**	**GDP per capita**	**Euclidean distance**	**Study pop.**	**Regression Weight**
Leitch and Hoyde 1996	Damage avoidance	Flood control	$26,700	1991	638,800	2.9085
Roberts and Leitch 1997	Damage avoidance	Flood control and habitat	$31,300	1867	106,406	2.8671
Roberts and Leitch 1997	Damage avoidance	Flood control	$31,300	1872	106,406	2.864
Leitch and Hoyde 1996	Damage avoidance	Flood control and habitat	$26,700	2191	638,800	2.7972
Costanza, Farber, and Maxwell 1989	Damage avoidance	Flood control	$23,800	1689	94,393	2.5647
Beran 1995	Stated pref.	Total value	$43,300	695	1,258,753	1.937
Petrolia and Kim 2011	Stated pref.	Total value	$36,800	1702	2,146,273	1.7274
Bauer, Cyr, and Swallow 2004	Stated pref.	Total value	$34,800	512	408,000	1.627
Bauer, Cyr, and Swallow 2004	Stated pref.	Total value	$34,800	512	408,000	1.627
Bauer, Cyr, and Swallow 2004	Stated pref.	Total value	$34,800	512	408,000	1.627

Table 23: Okefenokee NWR Stated Preference Flood control, Ten Highest Correspondence Observations

The top row of the table contains the policy site. Lower rows contain the ten observations with the highest predicted *correspondence weights*. Repeated studies (e.g., Roberts and Leitch 1997) represent unique value estimates that may share *correspondence attributes*.

Okefenokee NWR	Stated Pref.	Flood control	$36,400	0 km	3,531,251	N/A
Author-date	**Method**	**Service valued**	**GDP per capita**	**Euclidean distance**	**Study pop.**	**Regression Weight**
Leitch and Hoyde 1996	Damage avoidance	Flood control	$26,700	2217	638,800	3.0917
Roberts and Leitch 1997	Damage avoidance	Flood control	$31,300	2079	106,406	3.0558
Roberts and Leitch 1997	Damage avoidance	Flood control and habitat	$31,300	2082	106,406	3.0541
Leitch and Hoyde 1996	Damage avoidance	Flood control and habitat	$26,700	2453	638,800	2.953
Costanza, Farber, and Maxwell 1989	Damage avoidance	Flood control	$23,800	839	94,393	2.1704
MacDonald, Bergstrom, and Houston 1998	Stated pref.	Habitat	$29,700	319	2,793,672	1.5883
Beran 1995	Stated pref.	Total value	$43,300	326	1,258,753	1.4861
Petrolia and Kim 2011	Stated pref.	Total value	$36,800	870	2,146,273	1.422
Phaneuf and Herriges 1999	Revealed pref.	Recreation	$32,300	1596	630,561	1.3848
Phaneuf and Herriges 1999	Revealed pref.	Recreation	$32,300	1674	630,561	1.3639

Table 24: Sevilleta and Bosque del Apache NWRs Stated Preference Flood control, Ten Highest Correspondence Observations

The top row of the table contains the policy site. Lower rows contain the ten observations with the highest predicted *correspondence weights*. Repeated studies (e.g., Leitch and Hoyde 1996) represent unique value estimates that may share *correspondence attributes*.

Sevilleta and Bosque del Apache NWRs	Stated Pref.	Flood control	$32,900	0 km	3,531,251	N/A
Author-date	**Method**	**Service valued**	**GDP per capita**	**Euclidean distance**	**Study pop.**	**Regression Weight**
Leitch and Hoyde 1996	Damage avoidance	Flood control	$26,700	1604	638,800	3.1907
Leitch and Hoyde 1996	Damage avoidance	Flood control and habitat	$26,700	1714	638,800	3.123
Roberts and Leitch 1997	Damage avoidance	Flood control	$31,300	1563	106,406	3.0947
Roberts and Leitch 1997	Damage avoidance	Flood control and habitat	$31,300	1580	106,406	3.0848
Costanza, Farber, and Maxwell 1989	Damage avoidance	Flood control	$23,800	1600	94,393	1.7147
Sanders, Walsh, and McKean 1991	Revealed pref.	Recreation	$29,600	566	2,889,964	1.3919
Sanders, Walsh, and Loomis 1990	Stated pref.	Total use value	$29,600	566	2,889,964	1.3919
Sanders, Walsh, and Loomis 1990	Stated pref.	Total exist. value	$29,600	566	2,889,964	1.3919
Sanders, Walsh, and Loomis 1990	Stated pref.	Total option value	$29,600	566	2,889,964	1.3919
Sanders, Walsh, and Loomis 1990	Stated pref.	Total bequest value	$29,600	566	2,889,964	1.3919

Table 25: OLS and PLWLS Forecasts for 4 NWRs, Annual dollars per 1000 Acres per 1000 People per Year

2010 US dollars per 1000 acres per 1000 people per year		**OLS**		**PLWLS**	
Site	Service Valued	Median $\exp(XB_i)$	Mean $\exp(XB_i)$	Median $\exp(XB_i)$	Mean $\exp(XB_i)$
Arrowwood NWR	WQ	$170	$520	$370	$450
	FC	$100	$310	$110	$130
Blackwater NWR	WQ	$720	$2,210	$290	$330
	FC	$420	$1,300	$490	$550
Okefenokee NWR	WQ	$160	$480	$80	$90
	FC	$90	$280	$180	$210
Sevilleta and Bosque del Apache NWRs	WQ	$320	$970	$810	$980
	FC	$190	$570	$210	$260

Table 26: OLS and PLWLS Aggregate Values for 4 NWRs

Aggregate annual value 2010 US dollars per refuge per year						
Site	**Service valued**	**acres**	**OLS median**	**OLS mean**	**PLWLS median**	**PLWLS mean**
Arrowwood NWR	WQ	4,595	2,767,000	8,508,000	6,043,000	7,229,000
	FC		1,626,000	5,000,000	1,830,000	2,165,000
Blackwater NWR	WQ	24,502	62,294,000	191,523,000	25,479,000	28,748,000
	FC		36,606,000	112,547,000	42,653,000	47,530,000
Okefenokee NWR	WQ	375,778	208,374,000	640,647,000	101,817,000	120,588,000
	FC		122,450,000	376,474,000	238,576,000	279,353,000
Sevilleta and Bosque del Apache NWRs	WQ	5,106	5,682,000	17,468,000	14,551,000	17,661,000
	FC		3,339,000	10,265,000	3,838,000	4,625,000

CHAPTER 6

DISCUSSION AND CONCLUSIONS

To facilitate a comparison of the many benefit transfers in this dissertation, I combine the values obtained from meta-analysis benefit transfer using a total of 4 wetland meta-analysis models in Tables 27 and 28. These tables contain results for both flood control/storm protection and water quality provisioning. These tables contain forecasted ecosystem service valuation results from simulated stated preference studies for each refuge that use the 5 meta-analysis models (the last two models are based on the novel data set from Chapter 4). These tables include a row of results that are the median value or point estimate. Additionally, the mean values are included below each row of median values. As mentioned above, an important aspect of evaluating each model is consideration of the magnitude of the model's error variance. The larger a model's error variance is, the larger the divide between the median and mean value will be. The PLWLS model has the smallest gap between median and mean estimates of WTP, implying that the PLWLS model is the most precise.

The high precision of the PLWLS estimator comes at the expense of introducing variability through the estimated *correspondence parameters*; this increased variability is not captured by conventional, robust standard errors. However, the alternative *ad hoc* process of choosing a sample that is thought to have high correspondence with the policy site also introduces variability that is not accounted for when estimating standard errors. Accordingly with the PLWLS and all other econometric models, it is important to keep in mind that all results are conditional on the sample and model specification, and in the case of reported standard errors estimated for the PLWLS regression parameters the values are conditional on the estimated *correspondence parameters* which effectively serve as a means for resampling the data.

For the flood control/storm protection benefit forecasts using the PLWLS method, the top observations (Tables 21-24) are values from primary studies that use a damage avoidance approach to quantifying flood control/storm protection benefits. Pre-testing with a variety of combinations of *correspondence attributes* indicated that valuation approach (e.g., stated preference, revealed preference, damage avoidance, or replacement cost) was generally not a determinant of correspondence in this data set. Accordingly, forecasting a stated preference valuation result with a substantial proportion of the information coming from damage avoidance studies is more appropriate than using a stated preference study that values a different service. This result is useful because little theoretical guidance is available for developing *ad hoc* resampling rules. The PLWLS model finds high correspondence among flood control/storm protection primary valuation studies regardless of the valuation approach used in those studies. An interpretation of this result is that the active use component of flood control/storm protection (i.e., as measured by a damage avoidance study) is useful for understanding total value (i.e., as measured by a stated preference study).

All meta-analysis benefit transfer results in this paper can be interpreted as coming from a stated preference study, with the exception of the estimates obtained from the meta-analysis model of Ghermandi et al. 2010, which lacks controls for study method. While I present these 5 models together in Tables 27 and 28, it is important to caution that for the existing meta-analysis studies only the results of the median calculations have been made with all necessary information. Another important difference lies in the inclusion of the population over which benefits are aggregated as a dependent variable. The additional assumptions (that estimated parameters have zero covariance) required to calculate the mean value of the dependent variable for the meta-analysis studies other than the novel meta-analysis in Chapter 4 potentially lead to substantial inaccuracies. Not without surprise, I find that the accuracy of mean values obtained from the Monte Carlo assessment of published meta-analysis studies is questionable, especially considering how tremendously large some of these estimates are. Ultimately, I conclude that too much information is lacking to perform adequate benefit transfer with the information provided in the publication of each published meta-analysis study when mean values are of interest.

Figures 20 and 21 provide a graphic comparison of the median values obtained from each of the 5 meta-analysis models. These figures compare the results for the average acre of wetlands. Both figures have a vertical axis that has been logarithmically transformed to allow all 5 models to fit in a single, compact figure. The graphic presentation of the results from all 5 meta-analysis models indicates what I believe is the most reliable strength of using meta-analysis models for evaluating ecosystem service values; that is, meta-analysis models appear to be most valid for ranking sites in terms of the relative value of ecosystem services. For example, Figure 20 demonstrates that all but the Woodward and Wui (2001) results indicate that the average acre of Blackwater NWR supports the greatest (by a substantial margin) value of flood control/storm protection services. The same 4 meta-analysis models also indicate that the average acre at Arrowwood NWR or Okefenokee NWR consistently provides flood control/storm protection services of lower value than the other refuges.

Figure 21 provides a less consistent ranking, relative to the flood control/storm protection rankings, of water quality related ecosystem service values among the 4 case study sites. For water quality related ecosystem services, the wetlands of the Sevilleta and Bosque del Apache NWRs are consistently ranked as among the most valuable and the wetlands of the Okefenokee NWR among the least valuable. However, the relative ranking of the wetlands of Arrowwood NWR and Blackwater NWR for providing water quality related services are less consistent across meta-analysis models. For both of these figures, it is important to note that for all models except the PLWLS model the relative ranking across refuges within a model (e.g., in the Brander, Florax, and Vermaat (2006) model) cannot change between the two services, a restriction of the single equation OLS approach when interaction terms are not included.

In comparison to the median per acre values from the Brander, Florax, and Vermaat (2006) meta-analysis, the novel meta-analysis data set from Chapter 4 indicates dramatically higher median estimates of ecosystem service flows, though a direct comparison of the models is hindered by the omission of a user population variable in the existing meta-analysis studies. The higher values obtained from the novel meta-analysis model may be due to modeling a larger user population, though this cannot be assessed due to the lack of a control for this variable in the existing meta-analysis studies. In contrast to the relative ranking of forecasted median values across meta-analysis models the mean values obtained from the PLWLS estimator and novel dataset are generally smaller relative to the forecasts from the existing meta-analysis studies. Due to the problems with estimating the mean of the dependent variable using the existing meta-analysis models, as described in Chapter 3 and due to the problematic exclusion of population as an explanatory variable, I believe the results from the novel data set developed in Chapter 4 are more conceptually appropriate for meta-analysis benefit transfer. Additionally, due to the exclusion of non-domestic studies from the data set and due to the use of the PLWLS estimator, I expect to have greater correspondence and therefore reduced transfer error with the PLWLS model and the domestic data set.

The mean of the PLWLS estimator is what I consider to be the best benefit forecast, conditional on the population count used during estimation. An important contribution of this chapter is the modeling and associated acknowledgement that the population over which welfare estimates are aggregated in primary valuation studies is essentially always a choice of the original analyst and dictated neither by the model chosen by the analyst nor the context of the site associated with the ecosystem services being valued. Studies that utilize an empirical approach to restricting the population over which benefits are aggregated are surprisingly rare; Sutherland and Walsh's (1985) study is the only domestic exception I encountered in the literature search. Clearly an important next step is to develop a more formal empirical means for choosing the population over which benefits are aggregated.

Essentially, this study follows the existing meta-analysis valuation literature in agreeing that method-related covariates can be difficult to assign for forecasting ecosystem service values when one is simply interested in knowing how an acre of wetlands impacts the welfare of society. I expect that the estimation of a single number that is free of method-related underpinnings is not the goal of meta-analysis benefit transfer, not anytime soon. Rather, I reiterate the argument that the best interpretation of the dependent variable obtained from a meta-analysis regression model is a simulated primary valuation study result. As ecosystem service valuation studies are typically conducted in the context of answering a research or policy question, one must choose a specific valuation method to simulate a primary valuation study. If interest lies in a value that is an average of results obtained from a variety of methods, simulating each valuation method by coding method-related dummy variables to 1 or 0 and then taking the average (perhaps with unequal weights) of the final forecasts is more amenable to validation with primary studies. I specifically caution against coding dummy variables for binary concepts as fractions (e.g., coding them at their sample means); if this caution is not followed, the resulting forecasts will be non-linear functions of multiple valuation methods that have no clear interpretation and no clear means for empirical validation.

The results of the regressions using the novel data set produce numbers with a magnitude most comparable to the values estimated by Woodward and Wui (2001). In comparison to the valuation forecast results obtained from the median of the Ghermandi et al. (2010) meta-analysis model, the Chapter 4 OLS values tend to be about 50 times higher. The mean values simulated for the Ghermandi et al. (2010) study are substantially closer to the PLWLS mean values obtained with the novel data set, though substantial differences exist. Because readers do not have access to information regarding even the average population in the data sets associated with the 3 previously published meta-analysis studies, one cannot confidently say that the majority of the variation in benefits is not due to a difference in the population of beneficiaries that was used to make the forecast. Generally, one can say very little about the population of beneficiaries in meta-analysis regressions omitting a user population variable.

An additional avenue for future research with the PLWLS estimator is the development of a maximum likelihood approach. Specification of an analytical likelihood function over multiple sub-population regression equations may prove difficult, but identifying the probabilistic structure that gives rise to multiple centered populations may increase the reliability of results. Ultimately, stronger mathematical foundations are expected to increase the applicability and confidence in the PLWLS model.

The estimates of correspondence parameters obtained with the PLWLS meta-analysis model provide the first quantification of the multi-dimensional concept of correspondence in the literature. Because transfer errors are one of the greatest concerns with benefit transfers, identifying and reducing sources of these errors is an important task in developing defensible value estimates using secondary data.

Because the PLWLS estimator relies more heavily on some observations than others, an assessment of the method-related heterogeneity of primary valuation studies that was not controlled for with explanatory variables may provide a useful means for establishing the ideal valuation study for criterion validation. It is important to note that only with a clear valuation question and context does the notion of a "correct" valuation method exist; the variety of methods in a meta-analysis benefit transfer database are useful for answering specific but questions, but vague questions merit vague answers.

The majority of uncertainty regarding the "correct" method does not concern the broad categories of methods controlled for in meta-analysis models (e.g. TCM vs. CVM), but rather the specific details one might wish to implement when designing a valuation study of a chosen method. For example, CVM value estimates can be sensitive to the phrasing of a valuation question, which is a useful feature of the method, because the specific phrasing of the valuation question can elicit a response that is conceptually valid for evaluating a specific policy application. With meta-analysis models that only account for broad valuation methods, generalization errors are unlikely to become small in benefit transfer applications without sacrificing other aspects of correspondence. One potentially useful aspect of the PLWLS estimator is that the primary value estimates with the largest estimated *correspondence weights* can indicate the specific unmodeled characteristics of the benefit transfer forecast. In particular, if the top weighted observations were designed for similar policy applications, the benefit transfer forecast can be thought of as being relevant to the same type of policy application. This information can be used to construct validation studies with similar features to the highest weighted studies. With evenly weighted regression models, a benefit transfer analyst cannot point to the unmodeled features of a specific study and infer that these same features characterize the benefit transfer forecast.

I expect that for transfers with a large number of heterogeneous (both modeled and ignored heterogeneity) observations that are characterized by the highest estimated *correspondence weights*, generalization error is likely to be large. Similarly, for transfers that have relatively homogeneous observations that share the highest estimated *correspondence weights*, I hypothesize that transfer error will be relatively low. Primary valuation studies designed for testing criterion validity are needed to test the usefulness of this hypothesis in anticipating benefit transfer forecast error and validity.

Because formal mathematical analysis of the PLWLS estimator is limited at this point, conditions like the rank condition for OLS models have not been established. Consequently, it is not known how many correspondence parameters can be reliably estimated nor has it been determined how variations in the number of correspondence parameters interact with variations in the number of explanatory variables. Overall, the reduced variability relative to OLS indicated by the out-of-sample, jackknife simulated forecasts indicates that the PLWLS model is not overfitting the sample data. This observation suggests that the estimation method produces more accurate benefit transfers for a wider variety of policy applications than conventional regression models.

Management and Policy Implications

Benefit transfer is the practice of using existing economic valuation data from primary valuation studies to estimate economic values for sites and services that have not been valued. The use of benefit transfers for developing an understanding of the implications of land management strategies is appealing because of the relative simplicity associated with generating value estimates with benefit transfer models. However, by designing and conducting a primary valuation study, researchers can learn a great deal more about how ecosystem services affect populations than is revealed with a single numerical estimate. For example, in applying the contingent valuation method to estimate economic values, the process of using focus groups to develop a questionnaire that is understandable to respondents and suitable for answering management questions will provide much more information for policy makers and managers than can be obtained with a meta-analysis benefit transfer. As I mentioned in the introduction, the value estimates obtained with benefit transfers are based on broad generalizations. At small scales, local tastes and local ecosystem character may not be well-represented by available primary valuation studies, so the gathering and analyzing of new data will often be warranted. Generally, the best benefit transfer models and datasets are most useful when comparing ecosystem services at different sites and for different populations rather than for comparing the benefits of ecosystems to the benefits of built projects that entail the severe degradation of ecosystems.

Meta-analysis benefit transfers use a statistical model to make the most systematic use of available primary valuation studies for implementing benefit transfers Advancements in meta-analysis modeling precision like those offered by the PLWLS estimator developed in Chapter 5 are not a complete replacement for valuation studies that can incorporate complete information about local cultures, well-defined services, and the nuances of local ecosystems. Accordingly, these benefit transfer models can be used to answer similarly broad questions focusing on broadly defined services and populations. This broad-scale capacity means that organizations that are interested in comparing a large number of widely distributed landscapes and populations in order to answer questions that would require many separate primary valuation studies may benefit the most from using benefit transfer models. For example, non-profit organizations and government agencies focused on conservation at national and global scales may effectively apply benefit transfers in order to rank order potential land acquisitions in terms of ecosystem service benefits and real-estate costs. The PLWLS method is particularly apt for large collections of benefit transfers, like this example, as the estimator reduces the need for an analyst to resample from a larger dataset. In contrast to broad applications, local decision makers who are contemplating local policy decisions may have little to gain from benefit transfer models, as errors introduced by unmodeled heterogeneity can be substantial.

Even in broad applications of benefit transfer models, concerns have been voiced in the scientific literature about the appropriate methods and accuracy of benefit transfer value estimation. The advancements in the meta-analysis dataset compiled in Chapter 4 are useful for resolving some issues with the application of meta-analysis benefit transfer models. Generally, the interpretation of meta-analysis benefit transfers as a means for simulating hypothetical primary valuation results can help guide the forecasting process. For example, I include in the meta-regression a variable for the number of beneficiaries associated with a value estimate from each primary valuation study. This inclusion requires for meta-analysis benefit transfers that the practitioner decides who should be included in this population, just as occurs during the development of a primary valuation study. Viewing meta-analysis benefit transfers as a means for simulating primary valuation studies also clarifies the treatment of method variables; one should code method variables to indicate the type of primary valuation study that one would implement given sufficient time and resources.

By modeling the novel dataset with the PLWLS estimator, I am able to better understand and communicate the accuracy and performance of benefit transfer forecasts. In Chapter 3, I demonstrated that the use of existing model summaries leads to uncertainty about the precision of benefit transfers and thus uncertainty about the validity of using these existing model summaries to inform management and policy decisions. An important and related aspect of using meta-analysis benefit transfers for decision making is the choice of whether to use mean or median measures of the dependent variable. By developing a novel meta-analysis dataset, the accuracy of calculations of the mean are enhanced, reducing barriers to using the mean. Furthermore, by applying the PLWLS estimator rather than the OLS estimator to the dataset, the variability of the dependent variable is reduced, leading to a smaller difference between estimates of the median and mean. Because some policy decisions may hinge on whether the median or mean of the forecast is used, the PLWLS estimator is useful because it reduces the likelihood of this dilemma. The PLWLS estimator also offers the benefits of reduced forecast variability while also avoiding the introduction of bias from *ad hoc* resampling.

A final hazard associated with overstating the usefulness of benefit transfer models relates to the availability of funding for primary valuation studies. Overconfidence in benefit transfers may lead to a decline in funding for new primary valuation studies, and such a systematic pattern will threaten the quality of future benefit transfer models. I suggest that when benefit transfer models are used to answer a set of questions pertaining to broad landscapes and populations, that a comparison of some of the benefit transfers with novel primary valuation studies is essential for both validating the benefit transfers and improving the quality of future benefit transfers.

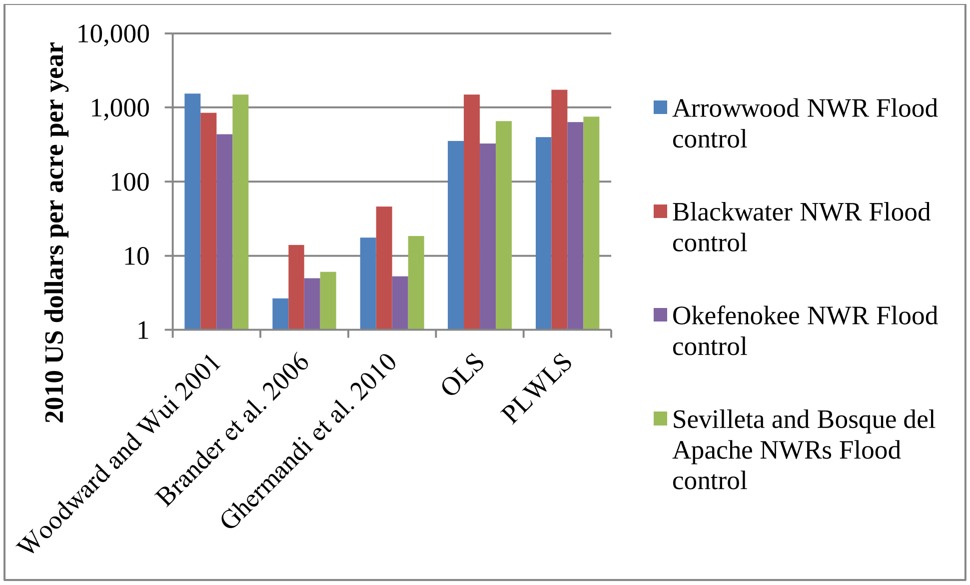


Figure 20: Meta-analysis Forecast Comparison for Flood Control, Annual Median Value per Acre

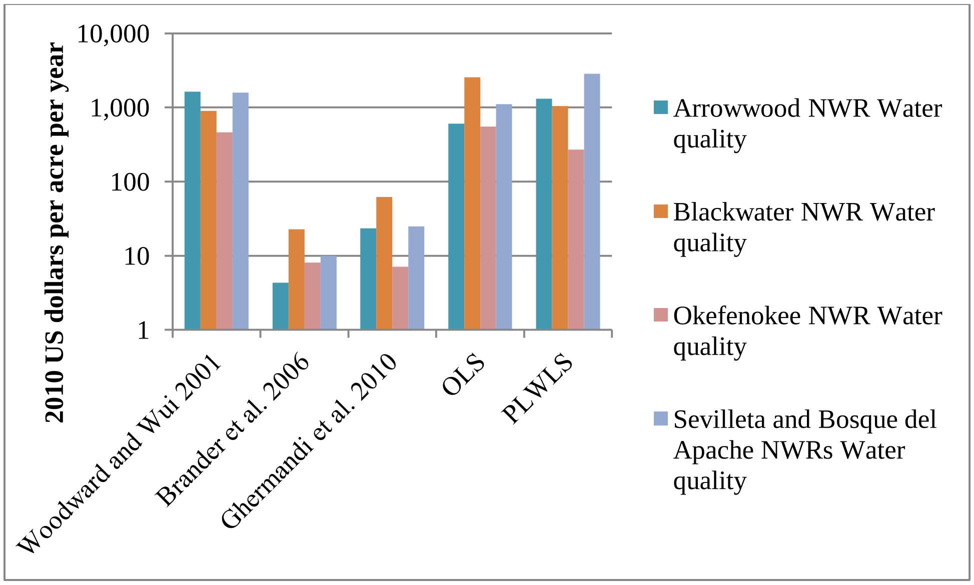


Figure 21: Meta-analysis Forecast Comparison for Water Quality, Annual Value per Acre

Table 27: Comparison of 5 Meta-analysis Models for 4 NWRs, Flood Control

2010 US dollars per year for NWR wetlands					
Arrowwood NWR					
Flood control					
	Woodward	Brander	Ghermandi	OLS	PLWLS
Median	7,043,776	12,177	80,325	1,626,115	1,830,089
Mean	553,429,493	4.780E+14	1,187,381	4,999,561	2,165,016
Blackwater NWR					
Flood control					
	Woodward	Brander	Ghermandi	OLS	PLWLS
Median	20,700,411	340,817	1,130,993	36,606,278	42,653,249
Mean	2,450,105,399	3.228E+15	20,781,772	112,546,849	47,529,586
Okefenokee NWR					
Flood control					
	Woodward	Brander	Ghermandi	OLS	PLWLS
Median	163,463,577	1,863,861	1,980,352	122,449,919	238,575,511
Mean	2.32E+10	1.78E+15	33,712,864	376,473,987	279,353,226
Sevilleta and Bosque del Apache NWRs					
Flood control					
	Woodward	Brander	Ghermandi	OLS	PLWLS
Median	7,594,007	30,992	94,150	3,338,566	3,837,809
Mean	603,359,646	3.73E+13	1,333,875	10,264,508	4,624,627

Table 28 Comparison of 5 Meta-analysis Models for 4 NWRs, Water Quality

2010 US dollars per year for NWR wetlands					
Arrowwood NWR					
Water quality					
	Woodward	Brander	Ghermandi	OLS	PLWLS
Median	7,471,872	19,898	107,897	2,767,365	6,043,124
Mean	600,924,752	1.102E+15	1,577,553	8,508,016	7,229,323
Blackwater NWR					
Water quality					
	Woodward	Brander	Ghermandi	OLS	PLWLS
Median	21,958,813	556,186	1,519,098	62,294,439	25,478,779
Mean	2,581,881,606	7.131E+15	27,620,452	191,523,451	28,747,552
Okefenokee NWR					
Water quality					
	Woodward	Brander	Ghermandi	OLS	PLWLS
Median	173,391,641	3,040,047	2,656,753	208,373,728	101,816,900
Mean	2.50E+10	3.52E+15	44,724,404	640,646,704	120,588,184
Sevilleta and Bosque del Apache NWRs					
Water quality					
	Woodward	Brander	Ghermandi	OLS	PLWLS
Median	8,055,619	50,598	126,419	5,681,530	14,550,536
Mean	655,118,693	7.85E+13	1,776,895	17,467,567	17,661,387

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