# National Wildlife Refuge Wetland Ecosystem Service Valuation Model, Phase 1 Report

**An Assessment of Ecosystem Services Associated with National Wildlife Refuges**

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# Executive Summary

The National Wildlife Refuge System's 150 million acres in over 500 refuges represent diverse landscapes with different capacities to provide ecosystem goods and services to society. Natural processes associated with management of national wildlife refuges provide benefits to local communities by sustaining production of specific goods and services that are useful to people. Estimated economic values of these services, such as those presented in this report, can be used to compare refuges in different locations and under different management, climatic, or socio-economic conditions. Our estimates of economic benefits from natural ecosystems serve as complements to economic impact analyses, such as the FWS's "Banking on Nature" studies (Carver and Caudill 2007).

This report presents the methods and results from Phase I of our research project. In this report we compare wetlands on four national wildlife refuges to illustrate how existing data can be used to estimate the average annual economic benefits of specific ecosystem services from different types of wetlands. The four sites are Arrowwood National Wildlife Refuge (NWR), North Dakota; Blackwater NWR, Maryland; Okefenokee NWR, Georgia; and Sevilleta and Bosque del Apache NWRs, New Mexico. These four sites were selected to contrast major types of wetlands in terms of physical and social parameters that influence the values of different ecosystem goods and services.

We present multiple approaches to assessing ecosystem services benefits. For each of the four refuges, we first consider a purely qualitative assessment of the relative magnitudes of different ecosystem service benefits provided by each refuge. This approach proves to be the most inclusive in terms of our ability to consider ecological data specific to the refuge, and

provides a useful tool for broad assessments and comparisons across refuges. However, it does not lead to quantitative estimates of ecosystem service benefits. For these estimates, we use two different benefit transfer techniques: (1) a meta-analysis benefit transfer to estimate the economic values of storm protection, water quality provisioning, and support for nursery and habitat for commercial fishing species; and (2) a point transfer approach to estimate the value of stored carbon. The results of the quantitative analysis are shown in Table A.

**Table A. Summary of estimated values of four wetland ecosystem services in four refuges (2010 US$).**

| Gross economic values per wetland acre per year | Storm protection | Water quality | Commercial fishing habitat | Carbon storage | 4 service aggregate |
|---|---|---|---|---|---|
| Arrowwood | $17 | $27 | $14 | $34 | $92 |
| Blackwater | $100 | $170 | $110 | $130 | $510 |
| Okefenokee | $70 | $120 | $0 | $140 | $330 |
| Sevilleta & Bosque del Apache | $47 | $80 | $0 | $14 | $141 |

| Gross economic values from refuge wetlands per year, values in thousands | Storm protection | Water quality | Commercial fishing habitat | Carbon storage | 4 service aggregate |
|---|---|---|---|---|---|
| Arrowwood | $80 | $120 | $60 | $160 | $420 |
| BlackWater | $2,000 | $4,000 | $3,000 | $3,100 | $12,100 |
| Okefenokee | $27,000 | $45,000 | $0 | $53,000 | $125,000 |
| Sevilleta & Bosque del Apache | $230 | $380 | $0 | $70 | $680 |

| Gross economic values, present value per wetland acre | Storm protection | Water quality | Commercial fishing habitat | Carbon storage | 4 service aggregate |
|---|---|---|---|---|---|
| Arrowwood | $540 | $880 | $460 | $1,100 | $2,980 |
| Blackwater | $3,000 | $6,000 | $3,000 | $4,200 | $16,200 |
| Okefenokee | $2,400 | $3,900 | $0 | $4,600 | $10,900 |
| Sevilleta & Bosque del Apache | $1,500 | $2,500 | $0 | $470 | $4,470 |

| Gross economic values, present value from refuge wetlands, values in millions | Storm protection | Water quality | Commercial fishing habitat | Carbon storage | 4 service aggregate |
|---|---|---|---|---|---|
| Arrowwood | $2 | $4 | $2 | $4 | $13 |
| Blackwater | $80 | $130 | $80 | $100 | $390 |
| Okefenokee | $890 | $1,500 | $0 | $1,700 | $4,090 |
| Sevilleta & Bosque del Apache | $7 | $12 | $0 | $2 | $22 |

Our results suggest that refuge size and the socio-demographic characteristics of the surrounding region are important determinates of the estimated per acre value of wetlands in providing ecosystem services. Consistent with economic theory, larger refuges in areas with lower population density tend to have lower per acre values. However, these interaction effects between wetland size, population size and preferences, and ecosystem service values need to be further studied.

Our results are an approximation of consumers' aggregate willingness to pay to obtain the service provided by the wetlands of a particular NWR. Decision makers can use these numbers to understand how a population might be impacted by a change in distribution of wetlands across a landscape. The most straightforward application of the method we follow concerns estimating the net economic value of a change in an ecosystem service due to a management action which changes a wetland from one type to another. This report represents Phase I of our efforts to estimate the ecosystem service benefits of the National Wildlife Refuge System. The primary focus of the second phase will be the development of a meta-analysis benefit transfer model specifically tailored toward wetlands in National Wildlife Refuges.

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## Section 1. Introduction

There has been a rapid increase in interest among natural resource managers to use ecosystem services for communicating with stakeholders and policy makers about the values natural processes contribute to society (Bergstrom and Randall 2010, Lamarque et al. 2011, Salles 2011). However, there are relatively few detailed studies providing broadly comparable measures of the benefits of ecosystem services. As experience increases among teams of ecologists, economists and managers, general agreements about concepts and working definitions are emerging (Heal et al. 2005, Brown et al. 2007, Bateman et al. 2010). The National Wildlife Refuge System's 150 million acres in over 500 refuges represent diverse landscapes that differ in their capacities to provide ecosystem goods and services to society, including clean water, clean air, flood mitigation, and recreation. High-quality information on the economic values of natural processes associated with management of national wildlife refuges can help explain how these areas provide benefits to local communities by sustaining production of specific goods and services that are useful to people. Estimated economic values can be used to compare refuges in different locations and under different management, climatic, or socio-economic conditions. These estimates of economic benefits from natural ecosystems serve as complements to economic impact analyses, such as the FWS's "Banking on Nature" studies (Carver and Caudill 2007).

The challenge lies in properly understanding and accounting for these important ecosystem goods and service. Without observations of market behavior, it is difficult to estimate the value of these goods and services to individuals who receive the benefits(Bergstrom and Randall 2010). Instead, non-market valuation techniques are used to estimate their magnitude. With limited conservation funds, identifying ecological and sociological variables that influence the value of ecosystem services supported by wetlands is necessary to ensure efficient conservation plans. We have organized our research activities into two phases. Phase I includes an initial analysis of ecosystem services at four refuges. The qualitative and quantitative methods and results of this Phase I component are presented in this report. Phase II includes future work that will build on and improve the initial case studies. A brief discussion of Phase II goals is provided at the end of this report.

**Goals and approach of Phase I.** Phase I research compares wetlands on four national wildlife refuges to illustrate how existing data can be used to estimate the average annual economic benefits of specific ecosystem services from different types of wetlands. The four sites are Arrowwood 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 along the Rio Grande within a single ecoregion and the availability of extensive data from the Sevilleta's Long Term Ecological Research (LTER) project. The choice of sites is intended to contrast major types of wetlands of the contiguous United States in terms of physical parameters (salinity, precipitation, temperature, distance to ocean, and distance to urban centers) and social parameters (income distribution, population density, and culture) which influence the values of different ecosystem goods and services and represent the range of diversity that could be compared quickly within the scope of our analysis.

To demonstrate both the advantages and the limitations of relying on existing data, we present multiple approaches to assessing the ecosystem services benefits. We first consider a purely qualitative assessment of the relative magnitudes of different ecosystem service benefits provided by each refuge. This approach proves to be the most inclusive in terms of our ability to consider ecological data specific to the refuge, and provides a useful tool for broad assessments and comparisons across refuges. However, it does not lead to quantitative estimates of the ecosystem service benefits. For these estimates, we first use a meta-analysis benefit transfer method to estimate the economic values of ecosystem services based on a meta-analysis (a statistical study of studies)published by Brander et al. (2006). The benefit of this approach is it allows us to estimate ecosystem service benefits without requiring the extensive data collection associated with a primary study. Examples of primary valuation techniques are the contingent valuation, hedonic price, and travel cost methods (Champ et al. 2003). With this approach, we estimate public benefits due to storm protection, water-quality provisioning, and support for nursery and habitat for commercial fishing species. A second quantitative method is used to incorporate carbon-storage data specific to our chosen sites. In this analysis we continue to rely on existing estimates of the value of stored carbon. We use a point transfer approach to obtain a per-unit value of stored carbon, allowing us to incorporate site-specific carbon storage data obtained from studies related to each refuge.

**Organization of this report.** The remainder of this report presents the methods and results of Phase I research. Section 2 presents a non-technical overview of the methods used to derive estimates of ecosystem service benefits for our qualitative and quantitative approaches. Section 3 describes the geospatial data used in the analysis. Section 4 presents the results of the analyses, organized by refuge. For each refuge (Arrowwood, Blackwater, Okefenokee, and Sevilleta & Bosque del Apache) we present the qualitative comparison of ecosystem service values, the quantitative estimates from the two benefit transfer approaches, and a summary of present values of the quantitative results. Section 5 discusses key elements of the results. Section 6 identifies possible applications of results and areas for future research. The technical details of the analyses are included in the Appendices. Appendix A provides a general discussion of important concepts related to ecosystem services. Appendix B provides theoretical details of the meta-analysis benefit transfer model, estimated coefficients from Brander et al. (2006), and an example of the steps involved in producing a meta-analysis benefit transfer estimate. Appendix C provides a summary of the model and results used to estimate carbon storage benefits. Appendix D describes our geospatial data analysis.

# Section 2. Methods

## 2.1 Qualitative Method: Expert Judgment

While the primary purpose of our study is to provide quantitative estimates of ecosystem service flows, we first conducted a preliminary qualitative analysis. This qualitative approach serves as our first approximation of ecosystem service values and incorporates the greatest breadth of site specific information, including population density, income distributions, the prevalence of substitutes, the occurrence of festivals, and other details.

In addition to a broad literature review focusing on each site, we visited each site to tour the refuge and to meet with scientists and managers familiar with the biology and social features of each site. During these meetings, we gathered information about visitor demographics, demand for various activities, and timing of visitation throughout the year. We discussed management objectives, including how intensively management acts to control fire, support wildlife populations, and (for Blackwater) restore marshland. For each refuge, we obtained literature relating to conservation plans, long term planning, and fliers for visitors to ensure that we understood the spectrum of relevant features associated with the multiple roles filled by each refuge.

## 2.2 Quantitative Methods: Summary of Ecologic-Economic Model

The overall objective of this research is to develop an ecologic-economic simulation model that can be used to evaluate the economic value of ecosystem services (see Appendix A for definitions) supported by National Wildlife Refuges. The primary advantage of this model is that is provides a means for evaluating ecosystem services when primary data studies are not possible due to funding and(or) time constraints. The ecologic component of the model estimates: 1) acres of different types of wetlands land cover in a National Wildlife Refuge using existing geospatial data (see Section 2.5 and Appendix D); and 2) amount of carbon stored above and below ground within different wetlands land cover types using existing data on the amount of carbon stored in wetlands vegetation (see Appendix C). The economic component of the model estimates: 1) approximations of willingness to pay per acre for three ecosystem services (storm protection, water quality, and habitat and nursery support for commercial fishing species) provided by different types of wetlands using meta-analysis benefit transfer (see Section 2.3 and Appendix B); and 2) point benefit transfer for carbon storage using existing studies on willingness to pay to avoid climate change damages (see Section 2.4 and Appendix C).

The major components and linkages in the ecologic-economic model are illustrated in Diagram 1 below. We demonstrate this model through application to four case study sites: Arrowwood NWR, Blackwater NWR, Okefenokee NWR, and Sevilleta & Bosque del Apache NWRs. More detail on the components of the model is provided below.

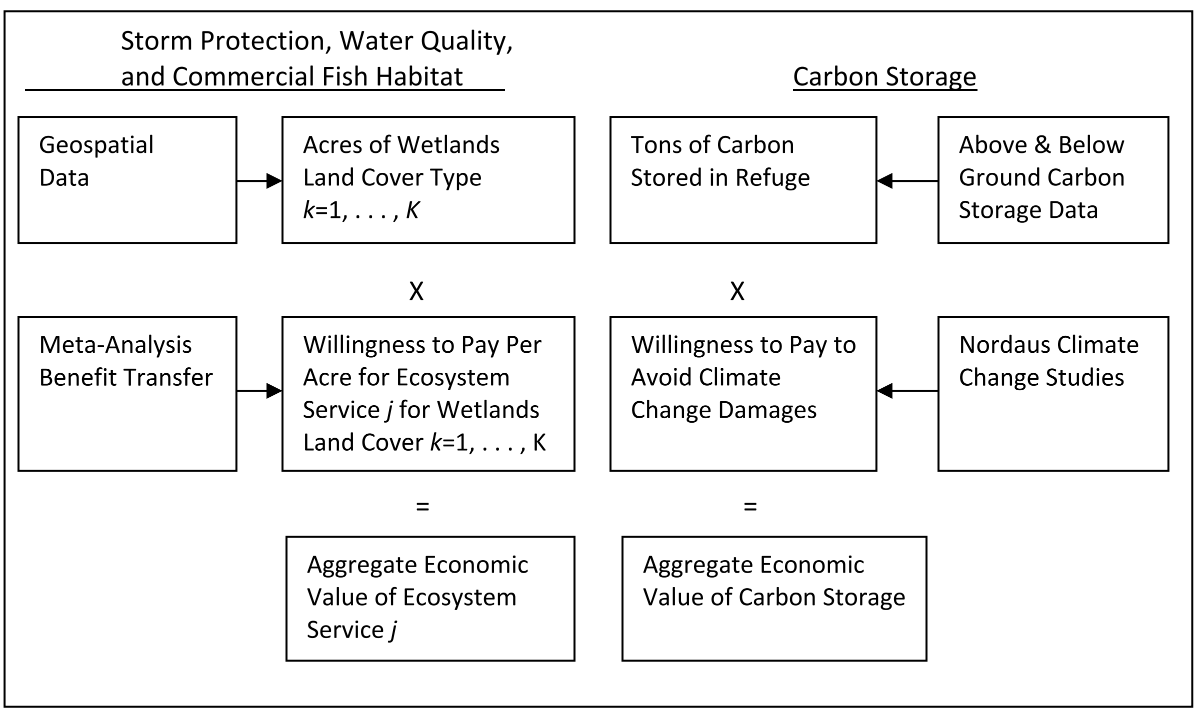


Diagram 1. Summary of Ecologic-Economic Model

## 2. 3 Quantitative Methods: Meta-Analysis Benefit Transfer

Non-market valuation methodologies rely on data from stated preference (Arrow et al. 1993) or revealed preference techniques (Bergstrom and Randall 2010), all of which involve extensive data collection, requiring much time and money. As more of these studies are completed, researchers have attempted to systematically define the manner in which estimates of ecosystem service value can vary in different locations (Potschin and Haines-Young 2011). In meta-analysis studies, variations in value estimates across studies are attributed to variations in the characteristics of the resource, such as the user population, the quality and quantity of ecosystem service flows, and the methods of estimation (Johnston et al. 2006). An application of meta-analysis results to estimate economic values is a meta-analysis benefit transfer (Bergstrom and Taylor 2006).

For our meta-analysis benefit transfer, we adapt the results of the meta-analysis from Brander et al. (2006) to estimate the economic values of storm protection, water-quality provisioning, and habitat and nursery support for commercial fishing species services supported by wetlands in the four selected NWRs. Their study is based on a large number of original studies and includes explanatory variables to characterize the landscape while also controlling for demographic variations, i.e. , local population density and national GDP. The inclusion of demographic variables is important as economic theory suggests they will significantly affect ecosystem service values. Brander et al. (2006)is also our prefered study because ittakes a more theoretically consistent approach to value estimation than Woodward and Wui (2001), an earlier wetland meta-analysis in the scientific literature (Bergstrom and Taylor 2006).

We focus on ecosystem goods and services rather than specifically valuing ecosystem structure or function (see appendix A for a discussion of ecosystem services) in order to avoid issues of double counting (Fu et al. 2011). We report quantitative economic value estimates based on primary studies that use the contingent valuation method (CVM). Many of the physical and social variables informing a qualitative valuation are excluded from consideration in our quantitative analysis because the existing meta-analysis models do not include these variables. Appendix B provides procedural details and values of explanatory variables as well as a discussion of the theory of ecosystem service values.

Our estimates of economic value are based on values transferred from primary studies which use the contingent valuation method. In an original CVM study, results are obtained from analysis of stated-preference survey responses (Champ et al. 2003, Farber et al. 2006) and include passive-use values that are not included in other available valuation techniques. Passive-use values include benefits such as preserving the option to use the resource in the future and knowing the resource exists for future generations. One potential concern regarding passive use values is that aggregation of passive use values can be more prone to double counting because survey respondents may consider benefits from multiple ecosystem services in their responses to questionnaires. Well conducted CVM studies are expected to minimize the occurrence of misstated preferences (Arrow, et. al. 1993).

The summation of estimated valuation results across multiple ecosystem services can produce biased results due to the possibility of path dependence in demand specification; accordingly, our results are based on the assumption of path independence (Just et al. 2005). While discussed conceptually, the meta-analysis literature related to ecosystem services does not include a practical treatment of joint estimation of multiple services. An intuitive example of the concern is as follows: two services when valued separately might sum to a larger value than if valued jointly because the services complement each other. Alternatively if the two services are mutual substitutes, the simultaneous valuation may be higher than the sum of independent valuation results. We do not believe this interaction to be an issue because the services we consider are expected to have only weak complement/substitute relationships.

Our analysis estimates the annual gross economic value of a stream of annual benefits of ecosystem services supported by the wetlands of each refuge. To allow for cost-benefit analysis, we estimate benefits as the present value (PV) of the annual flows of ecosystem services. Following the U.S. Office of Management and Budget guidelines, we employ a 3% discount rate over a horizon of 100 years (OMB 1992). The meta-analysis benefit transfer results are useful for estimating economic benefits of land on a surface area basis, providing an empirical means to estimate per acre economic values for water-quality provisioning, habitat and nursery support for commercial fishing species and flood protection. We illustrate another approach in estimating the value of wetlands in carbon storage.

## 2.4 Quantitative Methods: Carbon Storage Benefit Transfer

Research on climate change has identified anthropocentric emissions of greenhouse gases as a driver of global climate change (Houghton 1996), although the magnitude and impact are uncertain (Meehl et al. 2007). The value to human populations of averted climate change can be attributed to entities and processes which reduce the concentration and quantity of greenhouse gases in the atmosphere. While considerable uncertainty exists with regard to the value of storing carbon equivalent to a metric ton of carbon dioxide, existing estimates provide a range of possible prices under a range of scenarios. This range of prices is useful for long-term planning of options in managing ecosystem production.

We estimate the gross present value of carbon dioxide sequestration attributable to wetlands of our four selected NWRs through a price times quantity approach. While carbon storage values are generally viewed in present value terms, we also provide estimates of the annuitized annual value of climate regulation. Although carbon dioxide sequestration is often considered a supporting or intermediate service leading to the final service of climate regulation(Brown et al. 2007, Fisher et al. 2009), the standard approach in the literature is to consider carbon dioxide sequestration. The aggregate quantity of stored elemental carbon can be converted into its carbon dioxide equivalent ($CO_2e$), and then multiplied by an estimated value of a stored unit of carbon dioxide to obtain an economic value of the aggregate carbon store.

Simulation studies can help to estimate an efficient price for carbon storage that reflects the value for avoiding additional climate change with a reduction in carbon emitting economic activities. A series of dynamic macroeconomic models have been developed by Nordhaus, providing increasingly sophisticated estimates of the social cost of carbon. We apply the results of the latest revision of the Regional Integrated Model of Climate and Economy (RICE), discussed in greater detail in Appendix C. For the price component of the benefit estimate for carbon storage, we consider two estimates from the RICE 2011 model results: the 2015 estimated global social cost of carbon in the “business as usual” scenario ($13.02 per ton of $CO_2e$) and the 2015 estimated domestic social cost of carbon in this business as usual scenario ($1.10 per ton of $CO_2e$). These inflation-adjusted prices are relatively conservative estimates of the social cost of carbon compared to other estimated values, such as values estimated in the well-known Stern report (Stern 2008), which are more than 5 times greater than the higher value we consider.

An alternative approach would be to use market data on carbon credit prices from one of the artificial markets for stored carbon, which would provide an indirect indication of society’s willingness to pay for carbon storage. The Chicago Climate Exchange and European Climate Exchange provide a possible range of artificial market prices. Pricing on the Chicago Climate Exchange is much lower than on the European Exchange, averaging at about $2 per one hundred metric tons of carbon dioxide. However, these markets have well known limits that prevent them from fully functioning and so we do not consider them to be reliable measures of the social value of carbon storage.

We estimate carbon stocks for each refuge, following methodology generally consistent with the 2006 IPCC Guidelines for Greenhouse Gas Inventories. The use of carbon stocks allows for the estimation of gross values of carbon storage under the assumption of steady-state carbon stocks for the ecosystem under study. Carbon stocks are divided into above-ground and below-ground pools, which are each divided into living and dead carbon. While the value of carbon stored is independent of its storage in living or dead biomass, this distinction is often of interest to ecologists. A comparison of living and dead organic carbon over time is one way to consider loss of dead organic matter through the process of decomposition.

Unless otherwise specified, below-ground living biomass for forested wetlands is assumed to be 26% of above-ground living biomass, following Cairns et al. (1997). This assumption is due to limited scientific literature on below-ground living carbon in different types of ecosystems. The root to shoot ratios in wetland soils may be lower than for uplands due to increased availability of water. However, nutrient availability and other ecological variables are also relevant mediating factors in root development (Megonigal and Day 1992). Much of the scientific research on carbon storage has focused on upland forests (e. g. Cairns et al. 1997). Consequently where data on wetlands are not available, we use data for the closest relevant ecosystem.

We assume that stored carbon has reached a steady-state on the landscape level. Additional ecosystem data may establish non-zero net flows of carbon in a refuge. Valuation of carbon flows in addition to changes in stocks would allow for more precise accounting of carbon storage benefits. However, uncertainty in the temporal variability of carbon flows may lead to biased estimates. In addition to carbon dioxide, flows of nitrous oxide, methane, and several other gases have been identified as relevant to global climate regulation (IPCC 2006). Several methods have been explored to account for the impact of non-CO2 trace gases (Shine et al. 2005). Future research efforts can increase the accuracy of climate regulation value estimates by including non-CO2 greenhouse gases and accounting for the effects of temporal variation in relevant ecosystem processes (Hansen 2009). Other aspects of land use patterns and management practices are relevant to global climate regulation as these patterns relate to the complex structural and functional aspects of climate regulation (Marland et al. 2003). Moreover, as climate variability alters patterns of precipitation and carbon production, changes in inter-annual distributions of drought and wildfire will require further study of conditions when wetlands serve as sources for $CO_2$ and when they function as carbon sinks.

## 2.5 Quantitative Methods: Geospatial Data and Analysis

The US Fish and Wildlife Service maintains the National Wetlands Inventory (NWI) (USFWS 2011), a geospatial database following the wetland classification scheme developed by Cowardin et al. (1979). The Multi-Resolution Land Characteristics Consortium, a cooperative arrangement of the USGS, US Forest Service, US EPA, NOAA, and others, maintains the National Land Cover Database (NLCD). The NLCD is a set of nation-wide land classification maps which offer a separate source of geospatial data useful for identifying wetlands and for identifying the upland context surrounding wetland sites(Xian et al. 2009). While fundamental differences in mapping products and procedures leads to differences in accuracy assessments (Scheller et al.

2011), we find NWI classifications to generally identify fewer types of wetlands in the set of NWRs we consider. Because the NWI is maintained by the U.S. Fish and Wildlife Service (FWS) and the NWI generally follows a more conservative approach, we report our results using NWI data.

Using GIS software, we identify NWI wetlands contained within refuge boundaries (i.e. , within FWS Cadastral Geodatabase acquired boundaries). From this set of polygons, we are able to identify the following broad wetland types on our four refuge properties: forested, scrub-shrub, emergent, and unvegetated wetlands, which are further classified as freshwater or estuarine/marine wetlands. We provide maps of each refuge with NLCD 2006 land cover to illustrate the upland and lowland context of each refuge. Appendix D provides for more details on our use of geospatial data.

For the estimation of economic values for habitat and nursery support for commercial fishing species, water-quality provisioning, and storm protection using meta-analysis benefit transfer, we designated all freshwater forested/shrub wetlands as woodland; all forested and non-forested estuarine and marine wetlands as salt/brackish marsh; and all freshwater, non-forested/shrub wetlands as fresh marsh. The wetland type characterized as “unvegetated sediment” is a classification used by Brander et al. (2006) that has a relatively high value. However following a conservative approach, we reclassify NWI wetlands with classes such as unconsolidated shore and unconsolidated bottom as fresh marsh or salt/brackish marsh for meta-analysis, due to the presence of limited vegetation (Cowardin et al. 1979). Wetland types containing mixed NWI classification codes are assigned their lower valued status (i.e. , fresh marsh rather than woodland), based on coefficient values in Brander et al. (2006).

We follow a slightly different classification approach for estimation of carbon storage values in order to maintain our conservative approach. We aggregate wetlands based on vegetative cover, not differentiating among wetlands with varying salinity. Consequently, we reclassify NWI wetlands for purposes of carbon-stock accounting as forested, scrub-shrub, emergent, or unvegetated wetlands. Appendix D details the precise mapping of NWI wetland classes used.

Population values for the meta-analysis benefit transfer are computed from the 2008 US Census Bureau Population Estimates Program at the county level (US Census Bureau 2008). We compute population density for a radius of 50 km around acquired lands of each refuge as in Brander et al. (2006). Because several studies used in the meta-analysis were conducted outside the US one of the explanatory variables included is income per capita. We calculate the GDP per capita measure of $47,300 using 2010 GDP from the U.S. Bureau of Economic Analysis National Income and Product Accounts Tables and the U.S. population in 2010 is obtained from the 2010 Census Briefs (BEA 2010, US Census Bureau 2010). These 2010 nominal GDP per capita values are deflated to 1995 dollars using the US BLS CPI Inflation Calculator for use as explanatory variables in our meta-analysis benefit transfer (BLS 2011).

# Section 3. Results

## 3.1 Arrowwood National Wildlife Refuge

**Qualitative Comparison of Ecosystem Service Values.** Figure 1 is a NLCD 2006 map of Arrowwood NWR and surrounding lands. Figure 2 (based on the U.S. Fish and Wildlife Service National Wetland Inventory dataset) shows that 29% of lands acquired by the FWS are wetlands. Figures 3a and 3b show that nearly all wetlands at Arrowwood NWR are emergent or freshwater-marsh wetland. Additionally, much of the wetlands in Arrowwood NWR constitute 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. These values could not be incorporated into the quantitative estimates because of a lack of study of the benefits derived from migratory waterfowl nesting habitat support in the meta-analysis literature. 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, leading to a potential divergence from the more common pothole wetlands in ecosystem structure and function, and a consequent divergence in the value of ecosystem services. The substitutability between riparian and pothole wetlands is expected to be greatest for certain services, such as hunting, wildlife observation, and carbon storage. We expect flood 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 storm protection benefits to Jamestown, South Dakota. The existence of the riparian wetlands and 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).

The refuge supports a modest commercial fishing enterprise due to a desire to manage carp populations migrating upstream from the James River Reservoir. Generally, the value of habitat and nursery support for commercial fishing species is expected to be low for Arrowwood NWR due to limited production and long distance from major commercial markets.

Table 1 contains a qualitative evaluation of wetlands of Arrowwood NWR. We 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. Carbon sequestration in Prairie Pothole Region soils has been shown to be a significant sink for carbon, with native wetlands storing more carbon on average than farmed wetlands (Gleason et al. 2011, Gleason et al. 2008) while emitting comparable amounts of potent greenhouse trace gases such as nitrous oxide and methane (Gleason et al. 2009). 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 (e. g. DesHarnais et al. 1994).

**Meta-Analysis Benefit Transfer Results.** Considerable quantification of potential ecosystem service flows exist in the scientific literature for wetlands in the Prairie Pothole Region. Gleason et al., in their 2008 peer review study of primarily the USDA's Conservation Reserve Program and Wetlands Reserve Programs provide accounting of potential ecosystem service flows for carbon storage, floodwater storage, nutrient cycling, as well as other potentially useful measurements relevant to estimation of ecosystem service flows. The important links to human welfare, i.e., user populations and net or gross economic values, are not quantified in the study of Gleason et al. (2008). Future efforts to model the economic role of refuge wetlands in the Prairie Pothole Region may incorporate Gleason et al.'s data and analysis.

Technical details of how the meta-analysis was completed (for all four sites) are included in Appendix B. Table 2 contains the results of our meta-analysis benefit transfer for FWS acquired wetlands of Arrowwood NWR. The meta-analysis benefit transfer results estimate a yearly flow of ecosystem function for habitat and nursery support for commercial fishing species, storm protection, and water-quality provisioning ecosystem service values for the average acre at Arrowwood NWR to be $58 per acre and an aggregate yearly flow of services valued at approximately $265,000. Water quality related services account for a large portion of the economic value of the services we consider with an annual estimated value of $27 per acre and an annual aggregate estimated value of approximately $125,000 per year. Storm protection service value estimates are $17 per acre per year and approximately $76,000 per year in aggregate. Value estimates for habitat and nursery support for commercial fishing species are $14 per acre per year and about $64,000 per year aggregated across all acquired wetlands.

**Carbon Sequestration Results.** For the purpose of carbon accounting, we identify approximately 4,570 acres of emergent marsh, 17 acres of unvegetated wetland and 7 acres of forested wetlands. Above-ground living biomass for woody wetlands is derived from data on hardwood forests of North Dakota, as reported in Haugen et al. (2006). Parcels within refuge boundaries identified as forested wetlands in the NWI were assigned above-ground living biomass based on a computed-average dry tons of living biomass per acre of 18. Following Cairns et al. (1997), we estimate below-ground living biomass at 26% of above-ground living biomass. The carbon content of biomass in woody wetlands is assumed to be 47%, based on carbon content analysis (Lamlom and Savidge 2003) of ash, elm, and cottonwood trees, common North Dakota riparian species.

The carbon content of wetland soils is computed based on prairie pothole data provided in Gleason et al. (2008). The soil carbon content of 58.78 metric tons per hectare was applied to all identified wetlands, assuming soil carbon stores on all wetlands of Arrowwood NWR are equivalent to average native prairie catchments measured in Gleason et al. (2008). Similarly, above-ground vegetation on emergent herbaceous wetlands was assumed to be represented by Gleason et al. ’s (2008) data on native prairie catchments, with a carbon content of 1.47 Mg/ha. We use a root-shoot ratio of 1 for above-ground living biomass to estimate below-ground living biomass for emergent wetlands, based on data reported in Figiel, Jr. et al. (1995). Unconsolidated bottom and unconsolidated shore subclasses in the NWI, which we reclassify as unvegetated wetlands are assumed to have only below-ground dead carbon.

Table 3 provides details of carbon stores for each carbon pool for each land cover. Above-ground dead and live carbon stores in emergent marshes are reported by Gleason et al. (2008) as an aggregate number, and above-ground dead carbon is not estimated to be 0. Figure 4 provides a comparison of carbon stored in each land cover for each refuge. The majority of carbon stocks are soil organic carbon, which explains the small difference between carbon in emergent marshes and unvegetated wetlands.

Based on the estimated global social cost of carbon of $13.02 per Mg CO2e (Nordhaus 2011), we estimate the annual[1] value of carbon sequestration services supporting climate regulation for Arrowwood NWR to be $34 per average acre and approximately $160,000 in aggregate. This is a conservative estimate due to under identification of forested wetlands and scrub-shrub wetlands; another indication of a conservative bias in our carbon sequestration benefit estimates is due to the use of a relatively low root-shoot ratio of 1 for emergent wetland vegetation. If we consider only the U.S. domestic social cost of carbon, $1.10 per ton, the average acre's annual contribution to the gross economic value of carbon storage is $3 which is about $14,000 aggregated over acquired wetlands of Arrowwood NWR.

**Present Value of Aggregate Services.** Following the US Office of Management and Budget guidelines, we apply a 3% discount rate to estimate the present value of the stream of services, assuming a 100 year horizon (OMB 1992). Present value results for total-use values for Arrowwood NWR are reported in Table 4. The present value of storm protection services aggregated across acquired wetland of Arrowwood NWR is estimated to be approximately $2.5

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[1]We estimate the average annual contribution to climate regulation from the present value of stored $CO_2$e using the simplifying assumption that the annual efficient price of carbon dioxide emission reductions remains constant through time.

million and $540 for the average acre. The present value of water-quality provisioning services in aggregate is estimated to be about $4 million and $900 for the average acre. Habitat and nursery support for commercial fishing species is expected to provide a present value of about $2 million across acquired wetlands of Arrowwood NWR and $460 per average acre. Carbon storage has a present value of $1,100 per average acre and about $5 million in aggregate when valued at the global social cost of carbon. The present value of the four valued ecosystem services for Arrowwood NWR acquired wetlands is estimated to be $13.9 million, or $3,000 for the average acre.

## 3.2 Blackwater National Wildlife Refuge

**Qualitative Comparison of Ecosystem Service Values.** 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. Figure 5 is an NLCD 2006 map of Blackwater NWR and the surrounding landscape. 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 the Blackwater NWR occur at a relatively intensive level, including the management of impoundments, and agricultural plots on certain lands. 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 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.

Table 5 contains our qualitative valuation of ecosystem service flows at Blackwater NWR. Figure 6 demonstrates the abundance of wetlands among the acquired lands of Blackwater NWR. Figures 7a and 7b detail the distribution of wetland types for our quantitative models. Depending on their locations, saltwater-brackish marshes are often of greater value than fresh marsh due in part to increased interception of coastal storm surge, and reduced decomposition rates of dead organic material submerged in salt water.

The Chesapeake Bay is in close proximity to large and relatively high-income populations, thus we expect all services to be relatively valuable. The contribution towards nursery and habitat support for commercial fishing species we expect to be moderate to high due to provisioning of considerable estuary habitat of high quality. Additionally, commercial trapping economic values for nutria and muskrat pelts are included in the estimate of the economic value of support for commercial fishing species (Brander et al. 2006). Refuge biologists indicate that the primary species pertinent to nursery and habitat support for commercial fishing species are blue crab and white perch and to a lesser degree striped bass. We expect the water quality benefits from the Refuge directly benefitting humans to be quite high due to the proximity of large populations and the significant amounts of agricultural inputs upstream from the refuge. Moderate forest cover and submerged peat are expected to support moderate benefits from climate regulation services through the storage of potential greenhouse gases. Finally, we expect flood control and storm protection benefits to be high, as the Blackwater NWR acts as a barrier to storm surges that might otherwise damage valuable inland properties, for example 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 et al. 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 et al. 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 et al. 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. Our 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.

**Meta-Analysis Benefit Transfer Results.** The results of our meta-analysis benefit transfer are shown in Table 6. Water-quality provisioning services at Blackwater NWR are estimated to provide an annual flow of $170 per acre, or about $4 million in aggregate for acquired wetlands. Storm protection services are valued at an annual rate of $104 per acre, or approximately $2.5 million in aggregate for acquired wetlands. We estimate the value of services supporting commercial fishing species nursery and habitat at an annual rate of $105 per acre, and approximately $2.5 million in aggregate, which are consistent with the large and productive ecosystems of the Chesapeake Bay Estuary. The aggregate gross economic value of services estimated with the meta-analysis benefit transfer are estimated to have a yearly value of $378 per acre and an estimated value of about $9 million aggregated across all acquired wetlands.

**Carbon Sequestration Results.** Wills et al. (2008) estimate the carbon content of peat deposits in Blackwater marshes to be 24 $kg/m^2$, which are applied to unvegetated wetlands and emergent marshes. Above-ground living and dead herbaceous biomass measures were obtained from Stevenson et al.'s (2002) study of marsh restoration. Because Stevenson et al. (2002) report values for marsh that has been burned and unburned, we identify the proportion of emergent marshes on acquired lands which have burned in the last four years (52.4%) using geospatial data obtained from Blackwater NWR refuge staff. We use average above-ground carbon concentration of 43% for *Scirpus olneyi* and *Spartina patens* from Curtis et al.'s (1989) study of the effects of carbon dioxide enrichment in the Chesapeake for above-ground carbon content. Curtis et al. (1990) figure 4 reports below-ground carbon content in *Spartina*, *Scirpus*, and mixed plots, which do not visibly differ from above-ground carbon of 43% reported in Curtis et al. (1999). We assume a root-shoot ratio of 1 for living herbaceous biomass. Methane emissions in Blackwater are expected to be relatively low due to low sulfur soils (Wills et al. 2008), implying that the disservice of methane emission is small in economic value.

We estimate forested wetland carbon from the work on New Jersey's Pine Barrens of Scheller et al. (2011). Living above- and below-ground carbon are disaggregated under the assumption that 20.5% of living biomass is below-ground (derived from a root-shoot ratio of 0.26); we apply the estimated above-ground biomass for forested wetlands of 4,588 g $C/m^2$, and below-ground, living biomass of 1612 g $C/m^2$. For the below-ground dead biomass of forested and scrub-shrub wetlands, we apply Scheller's et al.'s (2011) value of 100 Mg/ha. A combined living and dead scrub-shrub above-ground biomass estimate of 7,829 Kg/ha are taken from the work on New Jersey's Pine Barrens by Ehrenfeld and Gulick (1981); we assume 43% carbon content for scrub-shrub above-ground biomass.

Table 7 contains a summary of our estimated carbon stores by carbon pool and land cover. We estimate the annual[2] value of stored carbon based on the global social cost of carbon estimated by Nordhaus(2011)of \$13.02 per $MgCO_2e$ to be \$129 for the average acre, or about \$3 million annually for all refuge acres. The gross annual value of stored carbon for the average

[2]We estimate the average annual contribution to climate regulation from the present value of stored $CO_2e$ using the simplifying assumption that the annual efficient price of carbon dioxide emission reductions remains constant through time.

wetland acre based on the U.S. domestic social cost of carbon is $11, which aggregates over all wetlands of Blackwater NWR to $260,000. The majority of the value of stored carbon at Blackwater NWR can be attributed to large stocks of below-ground dead carbon.

**Present Value of Aggregate Services.** The results of our present value calculations can be found in Table 8. We estimate the partial present value over a 100 year horizon at a 3% discount to be nearly $392 million for acquired wetlands of Blackwater NWR, or $16,200 for the average acre. The bulk of the estimated value is due to water-quality provisioning services, valued at $6,000 per acre, or about $131 million in aggregate. Storm protection benefits are estimated to be $3,000 per acre, or about $80 million in aggregate for acquired wetlands. Habitat support for commercial fishing species is estimated to support $3,000 per acre or about $81 million aggregated across all acquired wetlands of Blackwater NWR. We estimate the present value of stored carbon based on the Nordhaus (2011) carbon price of $13.02 per $MgCO_2e$ to be $4,200 for the average acre, or approximately $100 million for all Blackwater NWR wetlands.

## 3.3 Okefenokee National Wildlife Refuge

**Qualitative Comparison of Ecosystem Service Values.** The Okefenokee National Wildlife Refuge (NWR) occupies approximately 400,000 acres, mostly in Southeast Georgia with a small area across the border in Florida. As can be seen in Figure 8, 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 9, approximately 94% of the four hundred thousand acres acquired by the US Fish and Wildlife Service are wetlands. 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.

Figures 10a and 10b shows the distribution of wetland types in the Okefenokee for carbon analysis and meta-analysis benefit transfer, respectively. The extensive coverage of woody plants suggesting a high value for carbon storage. Table 9 contains a qualitative estimate of the relative worth of several ecosystem services supported by wetlands of the Okefenokee NWR. We expect values from habitat and nursery support for commercial fishing species to be low due to the distance from commercial fishing sites. We expect moderate 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). Carbon sequestration is expected to be high due to the abundance of peat and extensive forested wetlands. 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.

**Meta-Analysis Benefit Transfer Results.** Our meta-analysis benefit transfer results for acquired wetlands of the Okefenokee NWR are reported in Table 10. Due to our expectation that the value of habitat and nursery support for commercial fishing species is low, we adjust the statistical estimates to zero, a conservative estimate of this value of habitat and nursery support for commercial fishing species provided by wetlands of the Okefenokee NWR ecosystem. We estimate that an annual flow of three services estimated by the meta-analysis benefit transfer for the average acre in the Okefenokee NWR contributes an annual value of $192 per acre, which aggregated across the extensive wetlands of the Refuge results in an estimated value of about $72 million. Our estimates suggest that the highest valued service among those considered is water-quality provisioning, with an estimated annual value of $119 per acre, or nearly $45 million in aggregate. The value transferred to the average Okefenokee NWR wetland acre for storm protection is estimated to be $73 which aggregates to about $27 million across the refuge.

**Carbon Sequestration Results.** We obtain above-ground living and dead biomass data from Schlesinger (1978) and Greening and Gerritsen (1987). Below-ground living biomass pools are assumed to be 26% as large as above-ground living biomass pools, loosely following methods advised in the Georgia Carbon Sequestration Registry (Siry et al. 2006) and consistent with values reported by Cairns et al. (2007). Due to low nutrient availability in the Okefenokee, we expect root-shoot ratios to be higher than average upland, high-nutrient ecosystems (McJannet et al. 1995), thus our estimate of living below-ground biomass is likely conservative. Below-ground biomass data and carbon content data as well as invaluable background information are obtained from Cohen et al. (1984). Table 11 contains our estimates of elemental carbon storage in the wetlands of the Okefenokee NWR. Due to the variability of carbon stocks as a result of periodic fires, the numbers below might be conceived as a multi-decadal average. Future research is needed to understand the relationship between the fire dynamic of the Okefenokee NWR and the temporal variation in ecosystem service flows. Hamilton (1984) provides analysis with more information on the role of distrubance in the Okefenokee NWR.

As can be seen qualitatively in Figure 4 and quantitatively in Table 11, below-ground dead biomass (i.e. , peat) and above-ground living biomass constitute the bulk of carbon stored in the ecosystem. Our estimate of below-ground dead biomass in wetlands of the Okefenokee NWR is prone to under-estimation, as the distribution of peat was assumed uniform across all acres of Okefenokee wetlands. We expect that uplands have little to no peat deposits. On the other hand, we expect that our estimate of above-ground living biomass is biased upwards. Biomass data in Schlesinger (1978) for bog ecosystems are based on a dense cypress stand, while NWI identification of forested wetlands likely includes many stands of lower densities. We expect, however, that due to conservative placement of mixed scrub-shrub/forested wetland polygons in the scrub-shrub category, that the conservative carbon storage values estimated for scrub-shrub land cover balance the possibly high values for carbon storage on forested land cover.

Table 11 contains our estimates of the economic value of carbon stored in the wetlands of the Okefenokee NWR, evaluated at a range of possible prices to reflect the uncertainty of the magnitude and distribution with respect to time and across populations of damages associated with climate change. For the estimated global social cost of carbon, we estimate that FWS acquired Okefenokee NWR wetlands store carbon with an annual[3] value of around $53 million, or $141 for the average wetland acre. If we consider the U.S. domestic social cost of carbon, then the average acre has an annual value of $12 which aggregates over all Okefenokee NWR wetlands to nearly $4.5 million. Extensive below-ground dead carbon as well as extensive forested and shrub land cover contribute to the high carbon content of the ecosystem.

**Present Value of Aggregate Services.** Table 12 contains our estimated values for the aggregate present value for 100 years of flows of selected ecosystem services for acquired wetlands of Okefenokee NWR, discounted at an annual rate of 3%. We estimate the aggregate gross present value of the four services to be approximately $4 billion, or $10,500 per acre on

[3]We estimate the average annual contribution to climate regulation from the present value of stored $CO_2e$ using the simplifying assumption that the annual efficient price of carbon dioxide emission reductions remains constant through time.

average. Water-quality provisioning benefits are predicted to be most valuable, with PV worth approximately $1.5 billion, or $3,900 for the average wetland acre. Storm protection services are predicted to provide a present value of $900 million for all acquired wetland and for the average wetland acre the present value is estimated to be $2,000. We maintain our conservative approach in predicting a zero value for habitat and nursery support for commercial fishing species at the Okefenokee NWR. Carbon storage benefits evaluated at the global social cost of carbon have a present value of $1.7 billion in the aggregate with $4,600 for the average wetland acre.

## 3.4 Sevilleta & Bosque del Apache NWRs

**Qualitative Comparison of Ecosystem Service Values.** The Bosque del Apache includes approximately 57,000 acres of acquired lands, matching approved acquisition boundaries. The Sevilleta is significantly larger, including approximately 228,000 acres of acquired lands, also with no private inholdings (USFWS 2009). However, as can be seen in Figure 11, 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. 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. Wetland valuation results are estimated and reported as an aggregated value across the two refuges. As can be seen in figure 12, the two refuge system contains only 2% wetlands by surface area, with non-woody emergent or unvegetated wetlands constituting the bulk of wetland area, depicted in figures 13a and 13b. Scrub-shrub land cover dominates the woody wetlands, with only 1% of wetlands identified by NWI data as forested wetlands. We expect the value of ecosystem services supported by the extensive uplands of Sevilleta and Bosque del Apache to be significant, and 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 during the spring and summer primarily with corn and alfalfa. Corn, in addition to other crops, serves both to draw migratory waterfowl from surrounding agricultural lands and also as a source of feed for migratory waterfowl.

Our study 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. Table 13 presents a qualitative analysis of the relative value of ecosystems services supported by wetlands of Sevilleta and Bosque del Apache NWRs. 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.

The value of habitat and nursery support for commercial fishing species is expected to be low due to the large distance from important commercial fisheries. We 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. Carbon Sequestration is expected to be moderate for wetlands, primarily due to durable carbon stocks in woody wetlands in riparian corridors of the Rio Grande. Finally, we expect 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.

**Meta-Analysis Benefit Transfer Results.** The results of our meta-analysis benefit transfer can be found in Table 14. We predict the value of three services supported by the combined wetlands of the Sevilleta and Bosque del Apache National Wildlife Refuges to be worth approximately $600,000 on an annual basis, or $122 for the average wetland acre per year. The value of habitat and nursery support for commercial fishing species is predicted to be zero due to our prior expectations regarding the small magnitude of this service along the Rio Grande River. The value of storm protection services for the two refuges is estimated to be worth approximately $230,000 per year, or $47 for the average acre. Water-quality provisioning benefits are predicted to be highest among the service considered with an estimated annual flow of services of $380,000, or $76 per year for the average acre.

**Carbon Sequestration.** The Sevilleta and Bosque del Apache NWRs are situated along the Rio Grande River, but due to arid conditions the extent of wetlands is limited to riparian areas. Numerous control structures along the Rio Grande have significantly altered the natural flow regime. Numerous scientific studies have examined the effects of modern hydrologic management on ecosystem structure and function (Glenn and Nagler 2005, Molles et al. 1998, Sher et al. 2002). The ecological consequences of near total elimination of the natural flood regime are under long-term investigation at the Sevilleta Long-Term Ecological Research (LTER) project.

Because cottonwood trees (*Populus deltoids*) require periodic flooding for germination, reduced flooding may lead to declining populations of cottonwoods (Glenn and Nagler 2005, Valett et al. 2005) and the associated loss of relatively durable carbon stored as wood. Periodic flooding is also expected based on studies of similar sites (Stromberg et al. 2010) to reduce tree populations and canopy height along flood scoured portions of the bank, leading to a decline in stores of carbon. However a decline in flooding may lead to temporarily increased carbon storage due to decreased transport of woody debris and forest litter, while increasing the likelihood of fire and reducing nutrient cycling efficiency (Ellis 1999, Ellis 2001).

The combined effects of human management of the waters of the Rio Grande and the uncertain effects of global climate change present a formidable challenge in predicting future carbon stocks and flows in the riparian belts of the Rio Grande running through the Sevilleta and Bosque del Apache NWRs. We present estimates of stored carbon based on limited studies of existing riparian vegetation. Table 15 contains our estimates of carbon storage. Ellis (1999) reports above-ground woody debris biomass per hectare for a variety of forested sites. The pool of above-ground dead biomass is assumed to be represented by these data, ranging from 13.7 Mg/ha to 38.8 Mg/ha; we use the lower number, following a conservative approach. In a different publication, Ellis (2001) reports herbaceous biomass estimates in the forested understory of 0.226Mg/ha; this measure is additive to above-ground living biomass in forested wetlands, which are discussed below. Báez et al. (2007) provide measures of herbaceous biomass for an upland Chihuahuan black grama ecosystem of about 65 g/$m^2$, which we assume represents herbaceous wetlands. We use carbon content data from Tibbets and Molles Jr. (2005) for litter of 46%.

To estimate soil carbon storage we use estimates from McCulley et al. (2004), which reports woodland drainage carbon concentrations at 0-10 cm and 10-20 cm depths of 2,230 and 2,011 grams carbon per square meter, respectively and for grasslands at 0-10 cm and 10-20 cm depths of 987 and 749 g C/$m^2$, respectively. Soil carbon at depths below 20cm is not considered. These soil carbon measures are consistent with estimates from arid ecosystems in other parts of the world (Tiessen et al. 1998).

We use the online US Forest Service Forest Inventory and Analysis Database tool, available at http://apps. fs. fed. us/fido/, to estimate the average carbon stored per acre in cottonwood trees in inventoried plots in Valencia and Socorro Counties, New Mexico. We estimate the average carbon in living tree biomass above and below ground to be 23.3 Mg/ha. We use the same database to estimate the carbon content of shrub-scrub landcover assuming understory carbon densities in surveyed plots are comparable to shrub-scrub carbon densities. We estimate above-ground living and dead shrub carbon to be 3.87 Mg/ha; we expect this estimate of shrub-scrub biomass to be conservative for shrub-scrub landcover and additive to cottonwood biomass for forested acres.

We find that carbon stocks contribute modestly to the economic benefits of the Sevilleta and Bosque del Apache NWRs. Evaluated at the global social cost of carbon, we estimate the annual[4] value of carbon storage to be about $71,000 in aggregate, or $14 per acre per year. Evaluated at the U.S. domestic social cost of carbon, we find that the wetlands of Sevilleta and Bosque del Apache NWRs support and annual value of carbon storage services of $6,000, which implies the average acre contributes an annual value of $1 in climate regulation services through the storage of carbon dioxide equivalents.

**Present Value of Aggregate Services.** Table 16 contains the results of present value calculation for our combined approach to valuing acquired wetlands of Sevilleta and Bosque del Apache NWRs. The four services considered in our study are estimated to provide a present value over a 100 year period at a 3% discount rate of approximately $22 million, with the average wetland acre contributing an estimated $4,470. Storm protection benefits are estimated to provide

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[4]We estimate the average annual contribution to climate regulation from the present value of stored $CO_2e$ using the simplifying assumption that the annual efficient price of carbon dioxide emission reductions remains constant through time.

benefits worth$8 million in aggregate and $1,500 per acre over a 100 year period. Water quality benefits are predicted to be $12 million in aggregate and $2,500 for the average acre. The present value of carbon storage benefits evaluated at the global social cost of carbon is estimated to be approximately $2 million aggregated across acquired wetlands of both refuges, which implies the average wetland acre supports $470 in present value benefits.

## Section 4. Discussion

It is self-evident that, everything else being equal, larger wetlands will provide greater ecosystem service benefits, but it is important to understand that the average per acre value is not necessarily the same across all wetlands. Figures 14 and 15 contain surface area comparisons across the refuges. The aggregate values we estimate are strongly influenced by the number of acres of wetlands. We report the results of yearly ecosystem service flows aggregated across each refuge’s acquired wetlands for comparison in Figures 16 and 17. Much of the wide range in aggregate value estimates can be attributed to the number of acquired wetland acres. We separate the two refuges with extensive wetlands from the two with fewer wetlands to facilitate the comparison of values differing by orders of magnitude. Figure 18 demonstrates the estimated value of an average acquired wetland acre. Although there can be wide variation in per acre values, variation in the value of a year’s ecosystem services for an average acre is substantially less than variation in the aggregate values. The additional sources of variation in our valuation results are due to the interaction of biophysical and population properties of a refuge’s wetlands with the coefficient estimates obtained by Brander et al.

(2006) (see Appendix B) as well as variation in carbon stocks reported in the scientific literature. All values reported pertain to National Wetlands Inventory identified wetlands classified as acquired in the FWS Cadastral Geodatabase.

The empirical and theoretical effects on value of wetland surface area are an important variable to consider in the valuation exercise. Meta-analysis studies (e. g. Brander et al. 2006, Moeltner and Woodward 2009, Woodward and Wui 2001) and production function studies (e. g. Richmond et al. 2007) have been used to estimate returns to scale for ecosystem services. Brander et al. (2006), the primary study used in this analysis, estimate decreasing returns to scale for all services considered in their meta-analysis. Consequently, larger wetlands such as at the Okefenokee tend to have a lower predicted per acre value while smaller wetlands such as Sevilleta and Bosque del Apache tend to have higher values per acre. However, because other variables are not held equal across the refuges considered, the effect of scale is mixed with effects of population and ecological variables. Consumer demand theory suggests that while populations experience welfare benefits from a large flow of diverse ecosystem services, the benefit from marginal increments in services or quantities of goods will be valued less than prior increments in the service, giving rise to the well-known downward sloping inverse demand curve (Mas-Colell et al. 1995). Accordingly, under some conditions we might expect larger wetlands to have a lower per acre value than smaller wetlands for some ecosystem services, holding other variables equal.

Future studies may refine our understanding of the relationships between ecosystem scale and the value of ecosystem services. If the relationship between land area and certain ecosystem service flows demonstrates increasing returns to scale, the diminishing rate of returns to surface area expansion may be lowered or even reversed in effect. For example, future meta-analysis studies can estimate scale effects conditional on particular services, rather than an unconditional scale effect representing an average scale effect for all services.

Demographic differences also play an important role in our estimated results. Blackwater is located in the highest population density area, contributing to a higher estimated wetland per acre value than the other refuges considered. Income has a significant and positive impact on wetland values, but GDP per capita does not vary across the studies in our sample, so all estimates are affected equally. In addition to moderately high population densities relative to the refuges considered, the Okefenokee contains significant amounts of woody wetlands, which are the highest valued type of wetland according to the statistical analysis of Brander et al. (2006). The moderately high per acre value of the Okefenokee wetlands and their large extent contribute to an extremely high value wetland ecosystem.

Population densities in the vicinity of Arrowwood NWR and Sevilleta and Bosque del Apache NWRs are quite low compared to the national average, leading to relatively low values for each service. At Arrowwood NWR the preponderance of wetlands are lower expected value emergent marshes, suggesting that a value estimated for Arrowwood NWR without accounting for these features would be higher than otherwise. In contrast, Blackwater is predicted to be of comparatively high value by our methodology due to relatively high population densities in a 50 km radius around the refuge and also due to the higher value attributed to salt/brackish marsh relative to fresh, emergent marsh land covers.

A comparison of our valuation results with the estimates of similar studies provides a useful context to consider the magnitude of our estimates. Ingraham and Foster (2008) develop a benefit transfer methodology for valuing National Wildlife Refuges, and focus on four services supported by wetland habitats. Three of the four services they consider are somewhat comparable to the ones considered in this study. Ingraham and Foster (2008) apply the results of multiple individual studies covering different services for different landcovers, and scale point estimates of value for a particular service on a particular land cover by the ratio of net primary productivity (NPP) of the site under study to the site for which the value was estimated. This methodology is not grounded in economic or statistical theory, but uses the intuitive notion that economic values vary with NPP.

Following their methodology, we compute a prediction of the value of the wetlands in the refuges considered in our study. Based on the same ecoregion classification as Ingraham and Foster (2008) used, we assign Sevilleta and Bosque del Apache NWRs to NPP group 1, Arrowwood NWR to NPP group 3, Blackwater NWR to NPP group 5 and Okefenokee NWR to NPP group 7.

Table 17 contains a comparison of our results with estimates based on the methodology of Ingraham and Foster (2008). Generally, the results of the two studies suggest value flows of unequal magnitude, with the exception of habitat provision. Because the Brander et al. (2006) meta-analysis only considers habitat and nursery benefits as they relate to commercial fishing and hunting, the results from our approach are not entirely comparable with estimates obtained following Ingraham and Foster's (2008) approach. For the other wetland services considered by Ingraham and Foster (2008), a similar problem of only partial overlap of the services considered prevents a detailed comparison of the results.

Overall, we find that our results are substantially lower and thus more conservative for all refuges. Blackwater NWR is the closest estimate between the two approaches with our result approximately 10% as large as the estimates obtained through Ingraham and Foster's (2008) method. The differences between our estimates and those produced with the methodology of Ingraham and Foster (2008) can be attributed to two separate sources of variation. First, the inclusion of a wider range of theoretically important economic variables in our approach, such as population density, can be expected to lead to more accurate results. Second, because the services valued are not directly comparable, the difference in value may be attributed to differences in the services. This second issue relates to "commodity consistency", an important issue in meta-analysis studies, discussed in Bergstrom and Taylor (2006). Generally, because the services valued by Ingraham and Foster (2008) are more inclusive, the finding that our results are generally lower is consistent with our expectations.

## Section 5. Applications and Limitations of Results

Our model of ecosystem services is a combination of ecologic and economic components. For three of the ecosystem services studied (storm protection, water quality, and habitat and nursery support for commercial fishing species), the economic component is based on the Brander et al. (2006) meta analysis results as described in this report. Because in their conclusions Brander et al. (2006) “urge caution” in applying their results for benefit transfer, we do so likewise. [5]

Our results are a measure of the wetland user’s aggregate willingness to pay to obtain the service provided by the wetlands of a particular NWR. An appropriate application of these numbers by decision makers is to better understand, especially in an ordinal (or ranking) sense, how a population might be impacted by a change in distribution of wetlands across a landscape (e.g., land cover change). Understanding the effect on the value of flows of a particular ecosystem service due to land cover changes requires an understanding of the value of ecosystem services before the change and after the change. Meta-analysis studies of economic benefits from wetland ecosystems will generally be useful for comparing economic values from different configurations of a wetland landscape. Questions relating to causality, such as the net ecosystem service value of a management activity that leads to the conversion of forested wetlands to emergent herbaceous wetlands are suitable for meta-analysis benefit transfers such as those employed here.

Generally, a researcher can use a particular ecosystem service meta-analysis to generate estimates of net economic value for land use changes that can be adequately described by explanatory variables used in the meta-analysis. The most straightforward application of the

[5] Brander et al. (2006) estimated an average out-of-sample benefit transfer error of 74% which they state is comparable to other transfer errors reported in the benefit transfer literature. In their conclusions, they state that, “Given the high costs of performing primary valuation studies, this level of transfer error may be acceptable in considering transferred values as input in wetland conservation decisions.” (Brander et al. 2006, pp. 245).

method we follow concerns estimating the net economic value of a change in an ecosystem service due to a management action which changes a wetland from one type to another (e. g. from a woodland wetland to a fresh marsh). For example, the meta-analysis benefit transfer results allow one to compare the economic value of water-quality provisioning attributable to a wetland landscape across various distributions of woodland and fresh marsh wetlands, which may be linked to management decisions such as fire control. Similarly a researcher may apply our carbon storage valuation method to consider the net effect on carbon storage benefits of a conversion of forested wetlands to emergent herbaceous wetlands.

Another appropriate use of our results by decision-makers is ranking of refuges and(or) ecosystem services studied in terms of relative economic value. For example, our results indicate that out of the four refuges in our study, Blackwater NWR ranks the highest in terms of economic value of ecosystem services per acre when storm protection, water quality, habitat and nursery support for commercial fishing species and carbon sequestration values are aggregated. If getting the “biggest bang for the buck” is the policy goal, this ranking implies that Blackwater NWR should perhaps receive priority for additional acquisition funding. Within Blackwater NWR, out of the four ecosystem services studied, water quality ranks the highest in terms of economic value of ecosystem services per acre. Again if getting the “biggest bang for the buck" is the goal, our results suggest that water quality should perhaps receive priority for additional management funding, time, and(or) effort. [6]

The use of our results more in a cardinal sense, such as in benefit-cost analysis is more tenuous. [7] For benefit-cost analysis, the preferred methodology is a primary data nonmarket valuation study such a contingent valuation study or a travel cost method study. [8] Our results, however, could be of use to decision-makers for scoping and prioritizing primary data studies.

We postulate for a number of reasons that our result provide a conservative estimate of National Wildlife Refuge benefits. First, we only consider benefits to local populations whereas National Wildlife Refuges provided benefits to the nation as a whole implying that ecosystem service benefits of the refuges should be aggregated over the national population, not just the local population. Second, because one of the objectives of this study was to assess ecosystem services beyond recreation, recreation services (also considered ecosystem goods or services) were not included in our estimates. Recreational values are likely to be quite high for at least three of our refuge sites (Blackwater NWR, Okefenokee NWR, and Sevilleta & Bosque del

[6] Technically, economic efficiency analysis should be conducted using marginal values, not average values. Ranking based on average values is a "second best" option. Using average values is also consistent with a conservative approach as the Brander et al. (2006) meta-analysis indicates that marginal values of wetlands services are greater than average values (see Appendix B).

[7] However, it should be noted that many federal agencies including the USDA Forest Service, US Army Corps of Engineers, and US Environmental Protection Agency use benefit transfer for benefit-cost analysis.

[8] The same is true for natural resource damage assessment. The courts have recognized contingent valuation as a valid valuation tool for natural resource damage assessment in legal court cases. The same cannot be said for benefit transfer. We therefore do not recommend the use of our results in natural resource damage assessment, except for perhaps initial scoping purposes.

Apache NWRs). Third, because of lack of data, our results also leave out other ecosystem services such as biodiversity protection, aesthetic values, and cultural values (plus potentially many more). Fourth, as described above, we compared our results to our knowledge the only other existing study of the economic value of ecosystem services supported by National Wildlife Refuges (Ingraham and Foster, 2008) and found that our methodology generates lower estimates as compared to the Ingraham and Foster (2008) methodology. Fifth, the ecologic component of our model follows a conservative approach for allocating wetlands in our case study sites to specific land cover classifications (see Appendix D). Sixth, Brander et al. (2006) state that their meta-analysis estimates under-predict the value of relatively “high-valued” wetlands from ecologic and economic perspectives. We feel that the wetlands in our study fall into this “high valued” category (with the exception of perhaps the small amount of riparian wetlands in the Sevilleta NWR). Seventh, following Stapler and Johnston (2009) we have chosen to not implement an approximate upwards correction to the estimates we report to account for the non-linear specification of the dependent variable to further reduce the possibility of over-prediction; this correction is discussed at length in Bockstael and Strand (1987).

Thus, for scoping purposes, our results may provide first-approximation, lower-bound estimates of the economic value of ecosystem services supported by the National Wildlife Refuges represented in our study. Thus, if a proposed project passes the benefit-cost test using these lower-bound estimates, it is also likely to pass the benefit-cost test using more precise primary data estimates of the economic value of ecosystem services. However, we can only know for sure if a primary data valuation study is completed.

Another application of our results is to help prioritize where primary data valuation studies should be completed. For example, if a scoping-type benefit-cost analysis of a proposed wetlands project using our results indicates that the project would likely pass a benefit-cost test, decision-makers may want to prioritize the refuge where that project will take place for a primary data study.

We emphasize that our methodology provides a general demonstration model (framework) for evaluating the economic value of ecosystem services supported by National Wildlife Refuges. The ecologic (e.g., land cover estimates) and economic (e.g., willingness to pay estimates) components of the model can and should be updated when more accurate information becomes available. For example, because of acquisition and(or) natural changes (e.g., fire, sea-level rise, subsidence) the amount and distribution of land cover in a refuge may change over time requiring updating of the ecologic component of the model. Also, if more accurate estimates of willingness to pay per acre become available, the economic component of the model can and should be updated. The primary objective of Phase II of this research project described below is to provide more accurate estimates of willingness to pay per acre through estimation of an original meta-analysis function tailored more to National Wildlife Refuges.

**Future work.** This report represents Phase I of our efforts to estimate the ecosystem service benefits of the National Wildlife Refuge System. The use of an existing meta-analysis with a broad geographic and economic focus provided a means for the comparison of wetland related ecosystem services in a variety of contexts. There are several ways for improving the confidence in our estimates through enhancing the meta-analysis benefit transfer model.

The primary focus of the Phase II of this research project is the development of techniques for estimating the economic value of ecosystem services supported by refuge wetlands will be the development of a meta-analysis benefit transfer model specifically tailored toward wetlands in National Wildlife Refuges. Meta-analysis development can focus on increasing the number of ecological and socio-economic variables, allowing for consideration of a broader set of theoretically important sources of variation in the value of ecosystem services. Another important component of our proposed Phase II research is the expansion of the database of primary valuation studies included in the statistical model. Specifically, our current meta-analysis can be improved by developing a unique statistical simulation model more suited to answering specific questions related to National Wildlife Refuge study sites. A broader list of ecosystem services to include in valuation will also increase the usefulness of results. However, an increased risk of possible double-counting must be weighed against the advantages of a more complete list of ecosystem services. As different specific goals are defined and a more robust data set is collected, additional comparisons among refuges can be made with greater precision and confidence.

Small sample econometrics techniques such as bootstrapping and the use of Bayesian statistical techniques such as hierarchical priors and multi-model averaging allow for a more robust approach to meta-analysis specification and inclusion of non-sample information (Leon-Gonzalez and Scarpa 2008; Moeltner 2007; Moeltner and Woodward 2009). Also, the use of a wider range of landscape variables supported by spatial datasets includes greater diversity of ecological and sociological information.

Options for improvement of the model include a focus on identifying opportunities for cost-effective expansion of the refuge system. For example, comparing values derived from adjacent lands and associated natural processes can be useful for planning purchases of different inholdings and/or adjacent areas. These estimated values can complement other economic measures related to options for purchases in different locations with distinct land uses and land cover and at different distances from population centers. A second example of future research areas relates to understanding the effects of climate on the frequency of floods, droughts, wild fires, and sea-level rise. This set of research questions could be considered as a distinct or complementary project to determine the effects of seasonal and long-term dynamics on values for both market and non-market goods and services (e.g., carbon sequestration, pollination, flood control).

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## Appendix A. Concepts in Ecosystem Services

We follow a growing body of literature in our conceptual model of ecosystem services in differentiating between ecosystem structure, ecosystem function/processes, and ecosystem good or service (e. g. Boyd and Banzhaf 2007, Fisheret al. 2009, Wallace 2007). Our focus is on the contribution of ecosystem services to human well-being, and how goods and services relate to underlying ecosystem structure, processes and functions. While one can measure the value of ecosystem structure, processes and functions, and ecosystem goods and services, the valuation of goods and services is the only direct measure regarding human well-being (Fisher et al. 2009).

Table 18 depicts the conceptual linkages in our model. The goal of distinguishing between intermediate aspects of an ecosystem as the object of valuation and final goods and services, directly affecting human well-being is often motivated by a desire to avoid double-counting. Careful consideration of the spatial extent of ecosystem services and a consistent definition are also important considerations in ecosystem accounting (Fu et al. 2011). The Millennium Ecosystem Assessment (MEA) uses a similarly broad and anthropocentric definition of ecosystem services. The MEA classification of ecosystem services (provisioning, regulating, cultural, and supporting services) (MEA 2005), however does not provide exclusive categories which can be summed without concerns of double counting, e. g. climate regulation and food provisioning services.

## Appendix B. Technical Description of Meta-Analysis Benefit Transfer

The procedure for predicting values from an estimated meta-analysis regression is detailed below. Several datasets are required for accurate estimation; we provide the empirical values used in prediction in table 19, and the geospatial appendix discusses the estimation of several of the empirical measures. Below we discuss the contents of the tables of results in greater detail. See appendix D for information on our analysis and use of geospatial data.

We suppose that wetland ecosystems support ecosystem services according to the following equation,

$$\varepsilon_{ij} = f_i(E_j),$$

$$\text{where } E_j = r_j\,(N).$$

In the above model we assume that over a set of ecosystem services, $j$=1, 2, …, $J$ for consumers $i$=1, 2, …, $I$, each consumer chooses consumption, $\varepsilon_{ij}$ , of ecosystem service $j$, from available potential ecosystem goods and services, $E_j$. Potential ecosystem goods and services arise from the ecosystem processes, or ecological/economic transformation function, $r_j(.\,)$ that has as its argument the ecosystem's natural capital. The provisioning of potential ecosystem goods and services, $E_j$ is distinguished from actual ecosystem services, $\varepsilon_{ij}$ , due to the defining feature of goods and services as they relate directly to human welfare or well-being (e. g. an individual's use of flood protection services depends on where an individual locates her assets and how an individual protects her assets in the context of flood prone areas).

Microeconomic theory suggests the use of a Hicksian welfare measure, such as Hicksian compensating surplus (HCS), aggregated over the population of beneficiaries to estimate the benefits and costs of a policy that leads to a rationed change in ecosystem service consumption, $\varepsilon_{ij}$ (Randall and Stoll 1980, Willig 1976). The choice of the Hicksian compensating measure also includes an individual's response to the change as other market purchases are adjusted in pursuit of constrained utility maximization; this is in contrast to Slutsky compensation where other market purchases are held fixed.

Hicksian compensating surplus is a measure of an individual's willingness to pay to obtain a change in the consumption of a service such that the individual has utility equivalent to the case without payment and without enjoyment of the new quantity of the ecosystem service. This change can be modeled with microeconomic theory according to the following function,

$$U_i(Q^0, \varepsilon_{i1}{}^0, \varepsilon_{i2}{}^0, \ldots, \varepsilon_{ij}{}^0, B_i)$$

$$= U_i(Q^1, \varepsilon_{i1}{}^0, \varepsilon_{i2}{}^0, \ldots, \varepsilon_{ij}{}^1, B_i\text{-}WTP_{ij})$$

$$= V_i(P_q, \varepsilon_{i1}{}^0, \varepsilon_{i2}{}^0, \ldots, \varepsilon_{ij}{}^0, B_i)$$

$$= V_i(P_q, \varepsilon_{i1}{}^0, \varepsilon_{i2}{}^0, \ldots, \varepsilon_{ij}{}^1, B_i - WTP_{ij})$$

$$= U_i^0.$$

Where $U_i$ is individual $i$'s neoclassical utility function, which takes as it's arguments quantities of market goods and services, $Q$, and quantities of $j$ non-market goods and services, $\varepsilon_{i1}$, $\varepsilon_{i2}$, …,

$\varepsilon_{ij.}$ The superscripts 0 and 1 indicate values before and after a change in non-market goods and services. Utility maximization subject to a budget constraint leads to the indirect utility function, *V*. The indirect utility function contains $P_q$, a vector of prices of conventional and non-rationed or priced ecosystem goods and services in the economy, $B_i$, individual *i*'s budget, and $\varepsilon_{i1}^0, \varepsilon_{i2}^0, \ldots, \varepsilon_{ij}^0$, quantities of ecosystem services rationed to individual *i*. We use the term rationed to describe non-market ecosystem goods and services because the individual has incomplete control over consumption of these goods and services. The consumer will have a positive willingness to pay for ecosystem service, *j*, $WTP_{ij}$, when the consumption of that service is such that $\varepsilon_{ij}^1 > \varepsilon_{ij}^0$. The Hicksian welfare measure, compensating surplus, allows for optimizing adjustments to market good purchases, *Q*, due to a change in the availability of ecosystem services. An example of an optimizing adjustment to a (substitute) market purchase due to a change in consumption of ecosystem services is as follows: a consumer may purchase fewer water filters when water-quality provisioning services provide a relatively greater quantity of quality water. In addition, the Hicksian measure of compensating surplus implies that the consumer does not have a legal right to the higher level of ecosystem services, $\varepsilon_{ij}^1$.

In order to aggregate measures of CS from multiple changes, we must make the assumption that the order in which a consumer faces changes to rationed goods and services does not impact the welfare measure. For goods which are weakly related, this assumption is most reasonable. This assumption is known as path independence, referring to the notion that the path the consumer takes does not impact the magnitude of the economic value of the good or service to the individual. The gross economic value for the set of consumers, {1, …, $I$}, over the set of services, {1, …, $J$}, is then the aggregate or sum, $\Sigma_i \Sigma_j WTP_{ij}$.

The Hicksian measure of compensating surplus implies that the consumer has a legal right to the initial level of utility. The appropriate compensation under this assumption is a measure of willingness to pay (WTP) compensation to obtain an increase in consumption of ecosystem services ($\varepsilon_{ij}^1 > \varepsilon_{ij}^0$) and willingness to accept(WTA) compensation for decreases in consumption ($\varepsilon_{ij}^0 > \varepsilon_{ij}^1$). Generally, our results are intended to estimate Hicksian compensating surplus or willingness to pay to obtain a higher quantity of ecosystem services.

If we were to consider the Hicksian compensating surplus measure of welfare loss due to a downward change in the consumption of ecosystem services provided by a NWR wetland (which might be due to a degradation in the structure and function of a NWR landscape) and consumers had a legal right to the greater value of ecosystem services prior to the change, then we would be interested in the consumers' willingness to accept compensation for their loss. Because willingness to pay is constrained by one's wealth and willingness to accept is not, willingness to pay is often reported as a conservative estimate of willingness to accept (Arrow et al. 1993).

The meta-analysis benefit transfer we perform using the estimated coefficients of Brander et al. (2006) produces approximations of per acre willingness to pay Hicksian compensating surplus measures aggregated across the local user population for each service. While the theoretically desirable meta-analysis would produce welfare measures for individual consumers conditional on relevant characteristics of the consumer and ecosystem, the data requirements to estimate non-aggregate welfare measures for each individual from the entire user population for each service creates a formidable data gathering and modeling challenge. Alternatively, with a user population's per acre willingness to pay as the dependent variable, a researcher can use meta-analysis benefit transfer techniques to estimate a user population's per acre willingness to pay for the provisioning of an ecosystem service conditional on explanatory variables identified in the original meta-analysis. The meta-analysis of per acre welfare measures aggregated over a user population from existing primary valuation studies, allows for a parsimonious model, albeit with more numerous, less plausible assumptions.

Table 19 contains an example of the calculations and data used for estimating economic benefits using a meta-analysis regression equation. Statistically, we are interested in the expected value of ecosystem services, conditional on variables related to the human population, the wetland site, the service, and features of primary valuation studies. Empirical measures, such as population density, the continent where the wetland is located, and the distribution of wetland classes across the landscape are relatively straightforward. Primary valuation variables include the valuation method and the type of value estimate (marginal or average value). We condition our estimates on the contingent valuation method due to the capture of Hicksian welfare measures by this method and inclusion of passive-use values, as well as because the parameter for the contingent valuation method is significant at a 95% confidence level, unlike the parameters for alternate valuation methods. We estimate only average values to maintain a conservative approach, as marginal value estimates are substantially higher than average values, further at a 95% confidence level, the null hypothesis that the marginal value coefficient is different from zero cannot be rejected.

We reproduce Table 3 from Brander et al. (2006) in Table 20, which contains a list of explanatory variables with estimated coefficients and standard errors. Coefficients were estimated by the linear regression technique, Ordinary Least Squares and standard errors for coefficients were estimated by the heteroskedasticity robust Huber-White method (White 1980, Wooldridge 2002). The use of a regression model implies a causal relationship between explanatory variables and the dependent variable, which in this case is the natural logarithm of aggregate willingness to pay per hectare.

Most of the variables in the regression of Brander et al. (2006) are binary or dummy variables, taking on values of zero and one only. A dummy variable such as “South America” with an estimated coefficient of 0.23 can be interpreted as follows, a wetland in South America provides exp(0.23)-1=25.9% or approximately 23% more valuable ecosystem services than an otherwise identical wetland in North America. Typically dummy variables are interpreted relative to a single omitted category, so in the case of a group of location variables a single location is omitted, which is North America for the Brander et al. (2006) meta-analysis.

Several of the explanatory variables below and the dependent variable are in logarithmic form, for these independent variables such as the log of GDP per capita with an estimated coefficient of 1. 16 can be interpreted as follows: a 1% increase in GDP per capita causes a 1. 16% increase in willingness to pay for ecosystem services supported by a hectare of wetlands, all else held equal. For explanatory variables in their logarithmic form, the coefficient is thus a unit-free measure known as an elasticity. Most of the variables are in a linear rather than logarithmic form. For continuous measures in linear form, such as latitude with an estimated coefficient of 0.03, the appropriate interpretation is that a 1 decimal degree increase in latitude causes about 0.03% increase in willingness to pay for the ecosystem service supported by a hectare of wetland.

## Appendix C. Economics of Climate Change and Technical Details

Since the late 1950's, scientists have developed increasingly sophisticated models to explain and predict weather and climate. From an initial understanding of climate as a stable system with numerous feedbacks to support a stable equilibrium, scientific understanding of global climate has gradually rejected the notion of climatic stability as normal in favor of a system capable of numerous and highly varied steady-states sensitive to non-linear or chaotic feedback effects.

We provide estimates of the economic value of carbon storage in wetland ecosystems based on results from the latest refinement of William Nordhaus's regional dynamic general equilibrium model, RICE 2011 (Nordhaus 2011). Results from previous iterations of RICE dynamic models can be found in several peer-review publications such as, Nordhaus (1993), Nordhaus and Yang (1996), Nordhaus (2001), and Nordhaus (2010). The Rice 2011 model simulates economic and climactic conditions for 12 regions using a discrete-time dynamic general equilibrium model based on the Ramsey (1928) macroeconomic model. The Ramsey model consists of utility maximizing consumers who consume and invest in capital which is rented to firms and profit maximizing firms which rent capital from consumers and produce goods for consumption or investment.

In the RICE 2011 model, described in Nordhaus (2011) a social planner has the objective of maximizing the net present value of an increasing, concave social welfare function with a choice of investments in reductions in greenhouse gases. The optimal results from an economy directed by a social planner diverge from the private market outcome except for the efficient case (*i.e.*, when external costs of carbon are optimally internalized) requiring total cooperation among all nations, allowing for cost efficient emissions abatement. Alternative scenarios, such as when regions act uncooperatively or nations follow a “business as usual” emissions path (i.e. with no greenhouse emission controls) lead to lower estimated social welfare.

Generally, an economically efficient output of carbon balances the early losses from decreased production due to averted greenhouse gas emissions with the later losses expected to occur as an increasing function of greenhouse gas emissions. Following conventional assumptions in traditional macroeconomic models, exogenous technology and population growth are assumed. The exogenous growth assumption is consistent with observations of increasing consumption since industrialization; future people are assumed to be relatively rich in comparison with people at present. A consequence of increasing wealth over time is that reductions in future consumption have a smaller effect on welfare than reductions in present consumption. Thus we can view resulting estimates of the social cost of carbon as conservative relative to the scenario where consumption is not assumed to grow over time.

The social cost of a small change in carbon emissions from optimal emission levels determined by the model is the present value in terms of utility of the change in consumption due to losses from decreased future consumption relative to the consumption path determined in the model. Figure 19, reproduced from Nordhaus 2011, depicts the deviation in emissions from the optimal path and the resulting decline in consumption. Society's willingness to pay to return to the optimal consumption and emission paths will be equal to the present value of lost consumption.

Carbon stocks in wetlands depend on a variety of factors including wetland management. Society's gross willingness to pay for carbon storage provided by a landscape is a function of the stock of carbon that could potentially be released. When we confine our attention to only a single ecosystem service, e. g. climate regulation, society's net willingness to pay can be calculated by subtracting the opportunity cost, which is the gross value of the ecosystem service that the alternate landscape (i.e. were it not a wildlife refuge) would support. Predicting the use of a landscape under alternate management poses an additional modeling problem, complicating the calculation of net benefits. Society's gross willingness to pay for atmospheric carbon reductions is equal to the benefits of reduced carbon, i.e. reduced future costs.

The social cost of carbon varies with several important assumptions. Generally, in any model with investments that do not pay off for many years, the choice of discount rate will have a profound effect on the results. Assuming a particular discount rate is chosen, the social cost of carbon depends on the trajectory of factors such as emissions, consumption, and technology change which will vary with the use of emissions controls, the extent of effective international agreements, and also with many other facets of the world economy. An important determinant of the social cost of carbon is the emissions trajectory used as the baseline. Two important baselines to consider are the optimal baseline and the business as usual baseline, and in the RICE 2011 model, the social cost of carbon is not sensitive to the choice of one of these two bases. Below in Table 24, we reproduce Table 2 from Nordhaus (2011); one can see in the first column of the first two rows that the social cost of carbon changes little between the two scenarios. Thus results are robust to whether we consider the social cost of carbon from an optimal baseline or from a business as usual baseline. Table 24 also provides estimate of the social cost of carbon under alternate scenarios and assumptions.

Under the business as usual scenario, Nordhaus (2011) also provides country specific estimates of the social cost of carbon. We reproduce in our Table 25, Nordhaus's (2011) Table 3. The social cost of carbon for the United States is substantially lower than for the world as a whole, suggesting that if U.S. policy makers consider only internal costs of carbon when constructing control instruments then the reduction in emissions from U.S. sources will be small. Clearly assumptions about international cooperation have a substantial impact on the trajectory of emissions and the resulting social cost of carbon.

## Appendix D. Technical Details of Geospatial Data and Analysis

We conduct our analysis of U.S. Fish and Wildlife Service (FWS) National Wildlife Refuge wetlands using the FWS Cadastral Geodatabase to define the boundaries of the sites included in our analysis. The FWS Cadastral Geodatabase contains geospatial data for land parcels acquired by, administered by, or of interest to the FWS. Special designations such as land designated as Wilderness are also included in the FWS Cadastral. The FWS realty program is the primary source of data. Datasets such as the FWS Cadastral can be displayed as a map in Geographic Information System (GIS) software such as ArcGis 10, a widely-used product of Environmental Systems Research Institute, Inc. (ESRI).

The FWS maintains the National Wetlands Inventory (NWI), a geospatial database containing maps along with status and trend information for wetland and deepwater habitats of the United States (Wilen and Bates 1995). We employ the NWI for the purpose of identifying wetlands within the boundaries of National Wildlife Refuges.

An alternative wetland mapping system, the National Land Cover Database (NLCD) was also considered as a source of identifying information for wetland extent and type. The NLCD datasets are compiled and maintained under the Multi-Resolution Land Characterization (MRLC) consortium, which includes cooperators such as USFWS, USFS, NOAA, BLM, and more (www.mrlc.gov). A comparison of the NLCD and NWI maps revealed complementary aspects of the two systems. Specifically, the NLCD data contains information about all land cover types, providing data on the upland context of wetland systems. On the other hand, the NWI provides far greater categorical resolution, employing the hierarchal wetland classification scheme of Cowardin et al. (1979). Accordingly, we employ the NLCD 2006 dataset for maps and qualitative analysis while we rely entirely on the NWI data for quantitative analysis.

We use the clip tool in ArcGis 10 to retain only the elements of the NWI which are included within refuge boundaries. We define refuge boundaries as those parcels listed in the FWS Cadastral as 'acquired'. NWI data are in polygon format, allowing for precise clipping without introducing round-off errors. On the other hand, NLCD datasets are in raster or grid format, which introduces error to the clip process as a refuge boundary may intersect many grids, leading to round-off error as the mapping program does not split grid cells when clipping a raster file to a specific boundary.

We employ separate and distinct techniques for valuing carbon sequestration and the other three services considered, and due to a desire to maintain a conservative approach in estimation we employ separate mapping rules for each of the two techniques. Table 21 contains the mapping of NWI categories to the categories used in our model. Table 22 contains surface area estimates for each of our categories. The meta-analysis of Brander et al. (2006) includes estimated coefficients for the broad wetland categories(in order of decreasing value), fresh marsh, salt-brackish marsh, unvegetated sediment, and woodland. Wetlands in the NWI with scrub-shrub or forest cover are placed in the woodland category for meta-analysis benefit transfer. Wetlands identified in the NWI as of the estuarine class without woody cover are placed in the salt-brackish marsh category for meta-analysis benefit transfer. Wetlands in the NWI classes, palustrine, lacustrine, and riparian with subclasses, unconsolidated bottom, aquatic bed, emergent marsh, or a mixture of scrub-shrub or forest and emergent marsh are placed in the fresh marsh category for meta-analysis benefit transfer. No wetlands in the NWI are mapped into the unvegetated sediment category for the meta-analysis benefit transfer to prevent overestimation error due to an inability to conclusively determine that any particular wetland is better characterized as unvegetated sediment than fresh marsh or salt-brackish marsh.

To estimate carbon stocks on land, we employ a somewhat different mapping to avoid overestimation of carbon stocks on lands with mixed land cover. All wetlands identified in the NWI as having a mixture of subclasses are assigned to the category expected to have less carbon. For example, a mixture of scrub-shrub and forest is classified as scrub-shrub and a mix of forest and emergent marsh is classified as emergent marsh. For the purpose of estimating carbon stocks, we do not distinguish among wetlands with varying salinity. NWI wetlands identified as having relatively little vegetation (e. g. unconsolidated shore and unconsolidated bottom) are assumed to have the same below-ground living carbon as emergent marsh and the remaining carbon pools, following a conservative approach are assumed to be empty.

The wetland type, R4USJ, which often represents intermittent streams and dry washes is not classified as a wetland due to a potential lack of hydric soils or support hydrophytes. The R4USJ (reclassified as R4SBJ, but not updated in NWI dataset) classification is seen only at Sevilleta & Bosque del Apache among the refuges in our analysis; this wetland type tends to occur only in the arid west (details available via the FWS Wetlands Code Interpreter, http://www.fws.gov/wetlands/Data/WetlandCodes.html).

To estimate the population density for a 50 km radius around each site, we use US Census Bureau county population estimates for the year 2008 and 2010 US Census TIGER/Line® Shapefiles surface area data to estimate average population density in the vicinity of each refuge. Because county population counts may include high population density cities which are not within a 50km radius of a refuge, we only use counties with population densities that are representative of the 50km radius around each site. An example of this procedure can be seen in our population density estimates for the Okefenokee NWR's vicinity. We exclude the three coastal counties (Camden, Duval, and Nassau), as the county-level population density estimate is not representative of the population densities in the part of each county near the Okefenokee. Table 22 contains population and geographic data used in our calculations of population density for a 50 km radius around each site.

# Figures

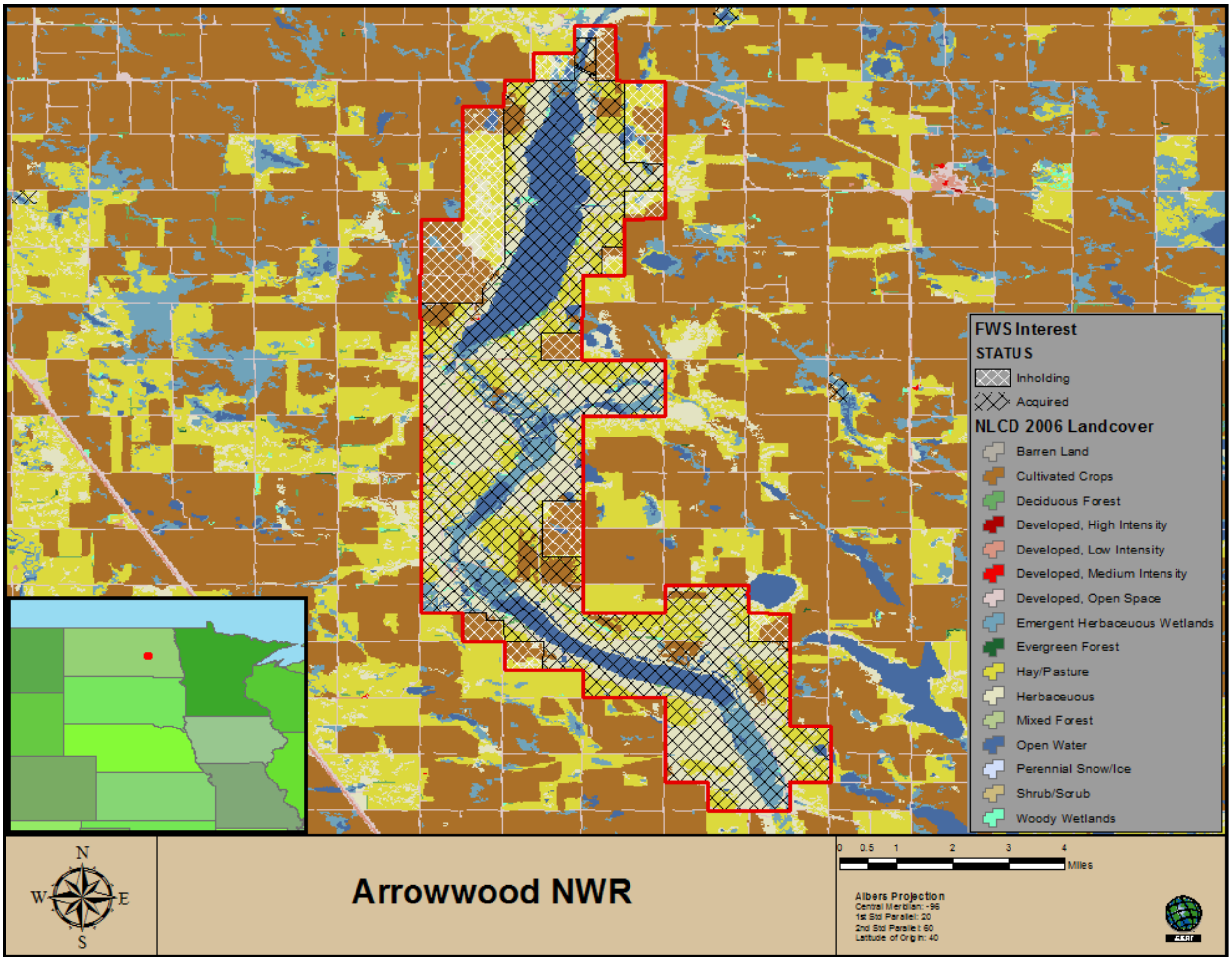


Figure 1. Arrowwood NWR, NLCD 2006 landcover.

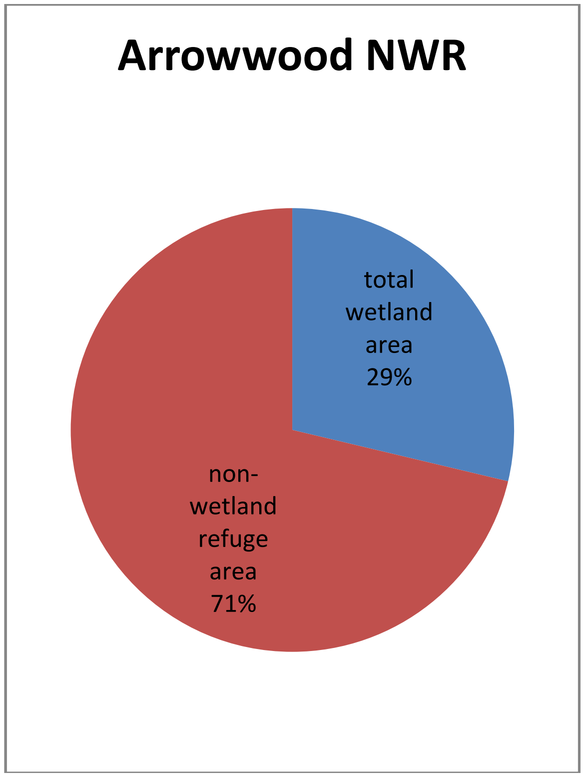


Figure 2. Proportion of Arrowwood NWR acquired acreage identified as NWI wetlands.

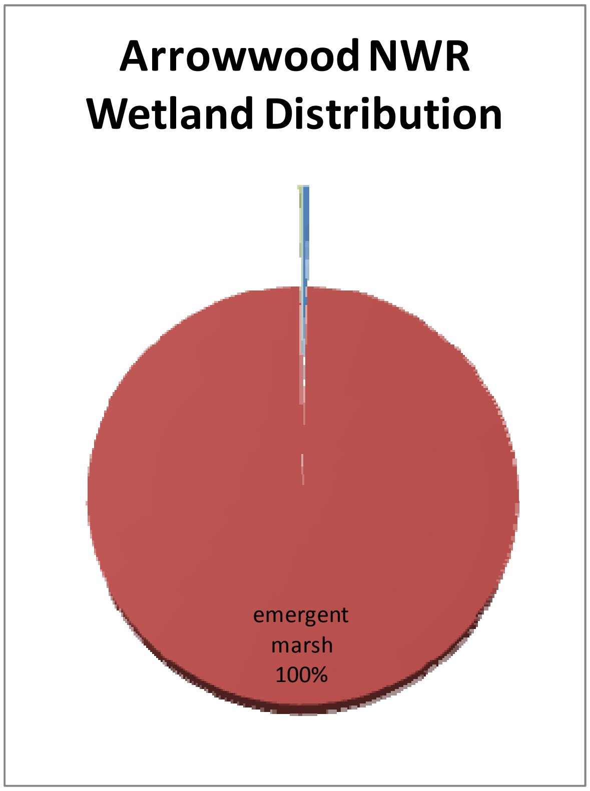


Figure 3a. Distribution of NWI wetlands at Arrowwood NWR for carbon storage analysis.

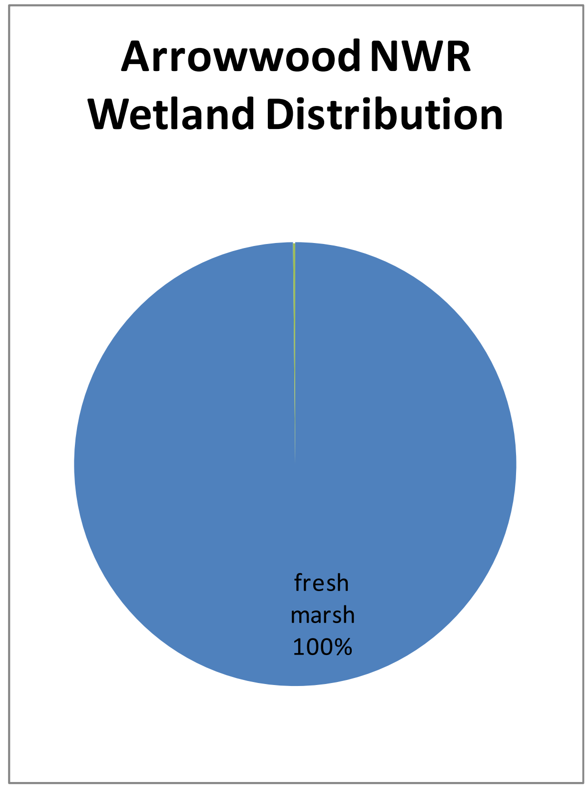


Figure 3b. Distribution of NWI wetlands at Arrowwood NWR for meta-analysis prediction.

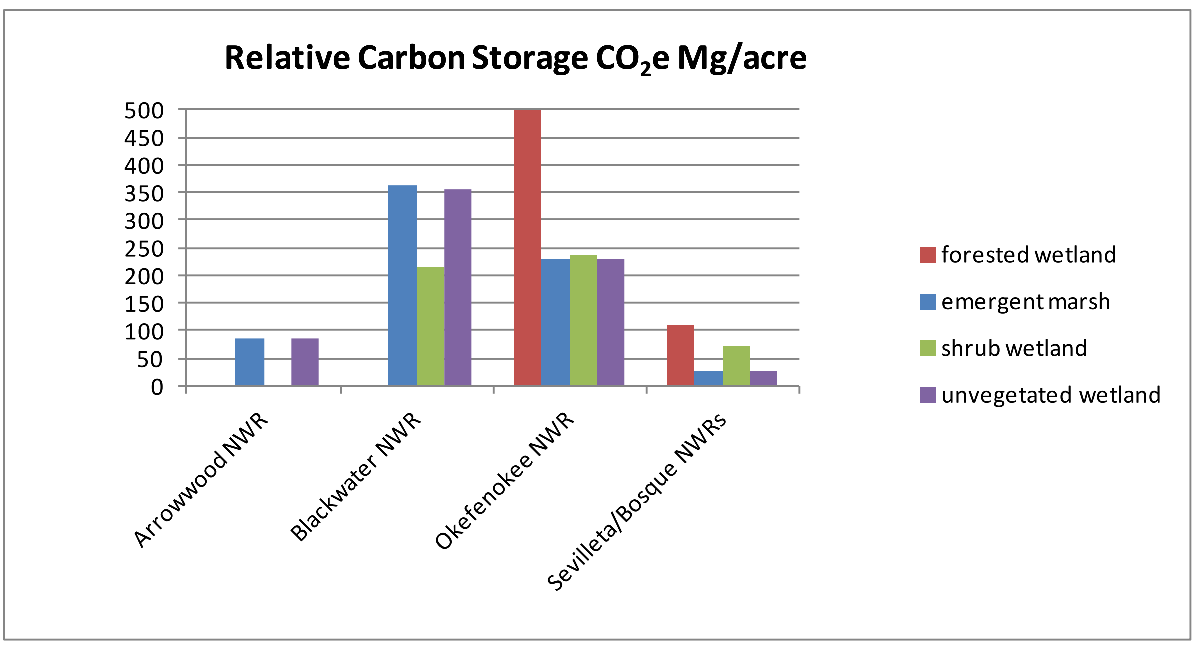


Figure 4. Relative carbon storage per acre by landcover and refuge.

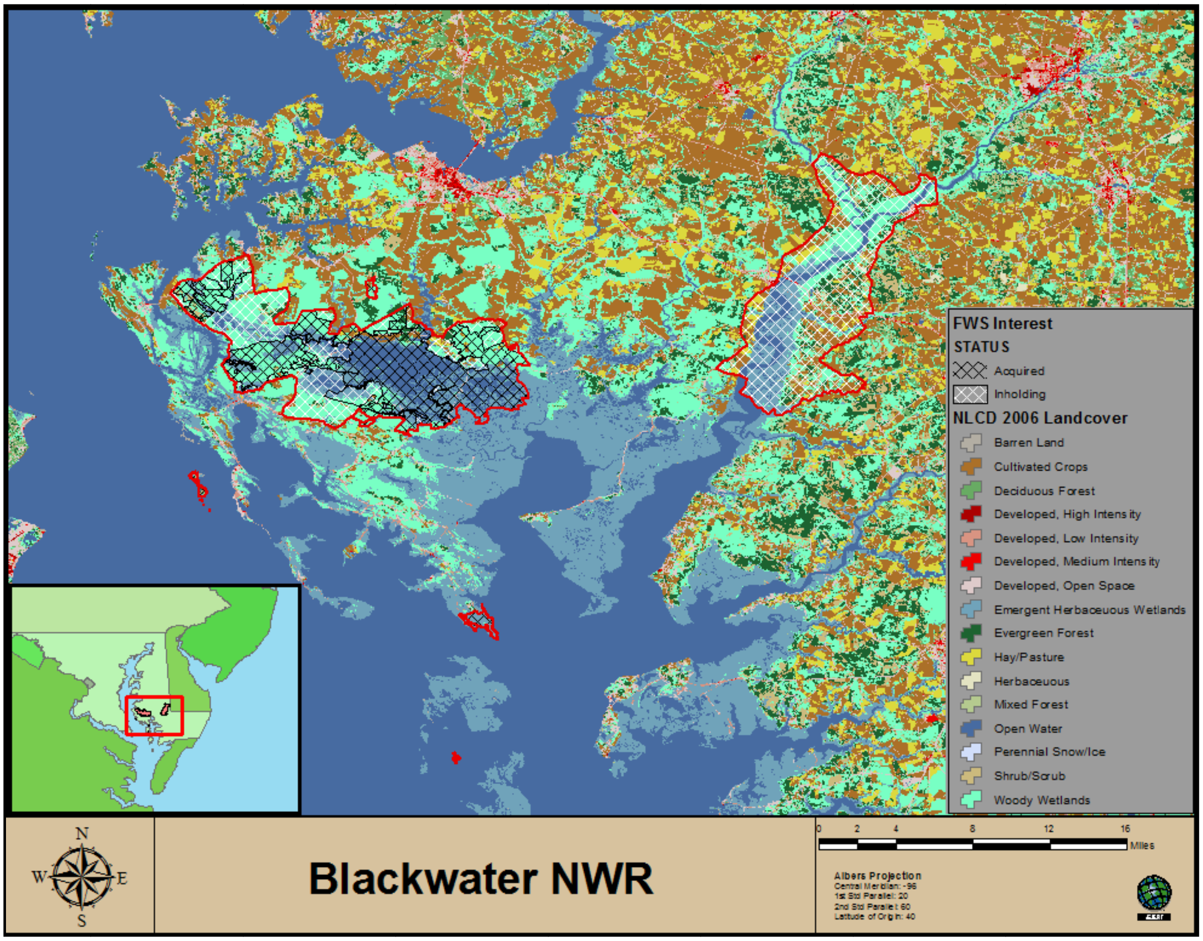

Figure 5. Blackwater NWR, NLCD 2006 landcover.

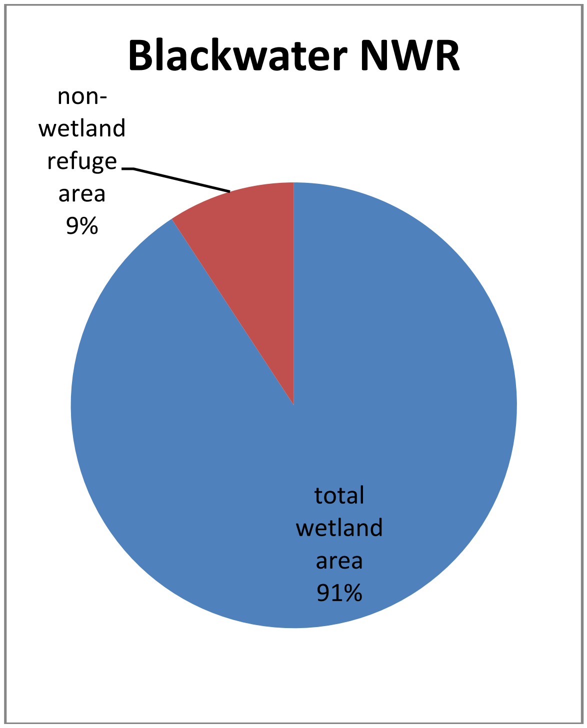


Figure 6. Proportion of Blackwater NWR acquired acreage identified as NWI wetlands.

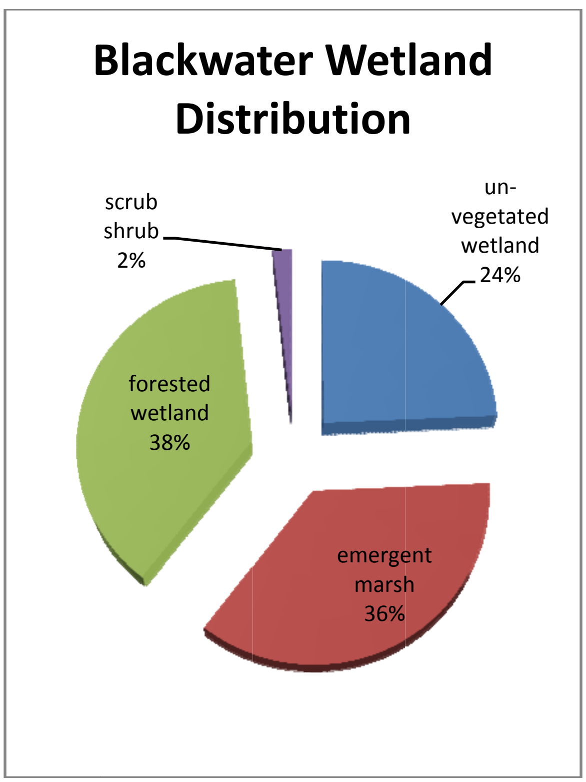


Figure 7a. Distribution of NWI wetlands at Blackwater NWR for carbon storage analysis.

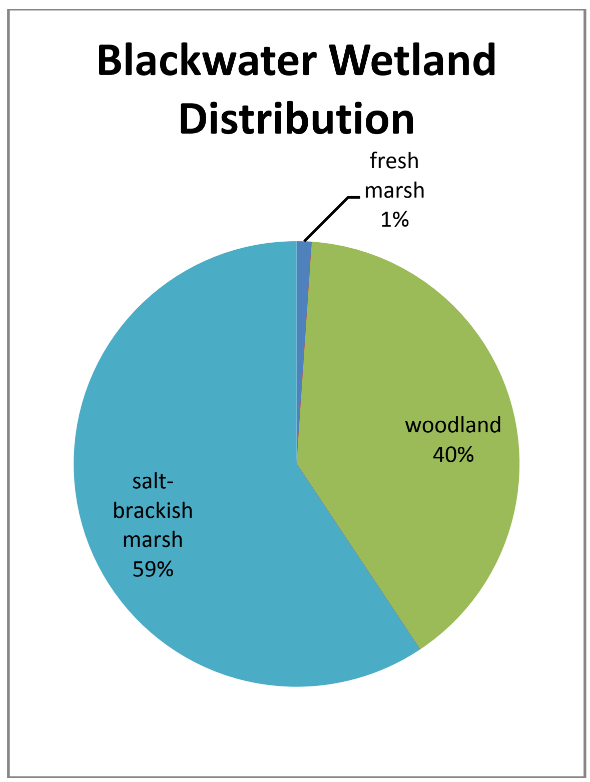


Figure 7b. Distribution of NWI wetlands at Blackwater NWR for meta-analysis prediction.

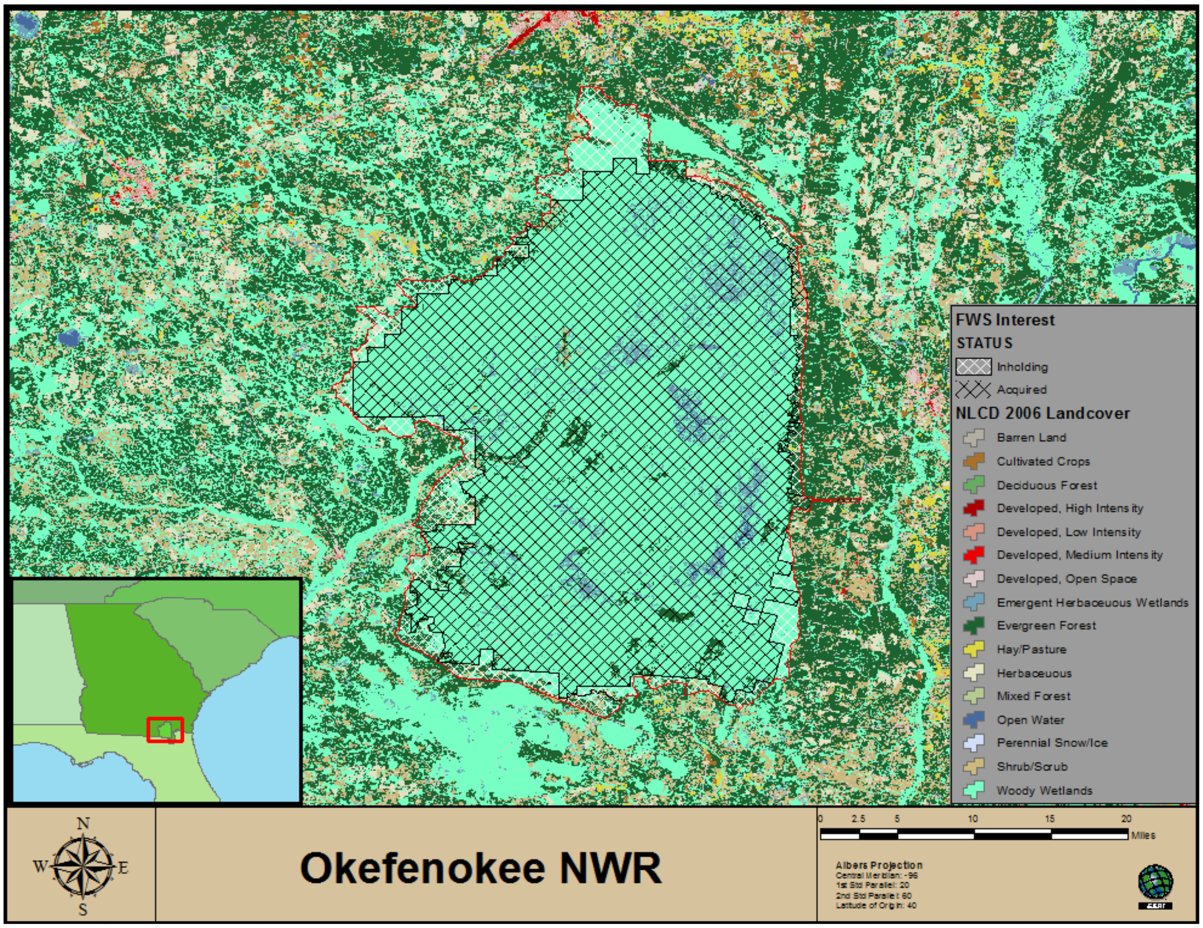


Figure 8. Okefenokee NWR, NLCD 2006 landcover.

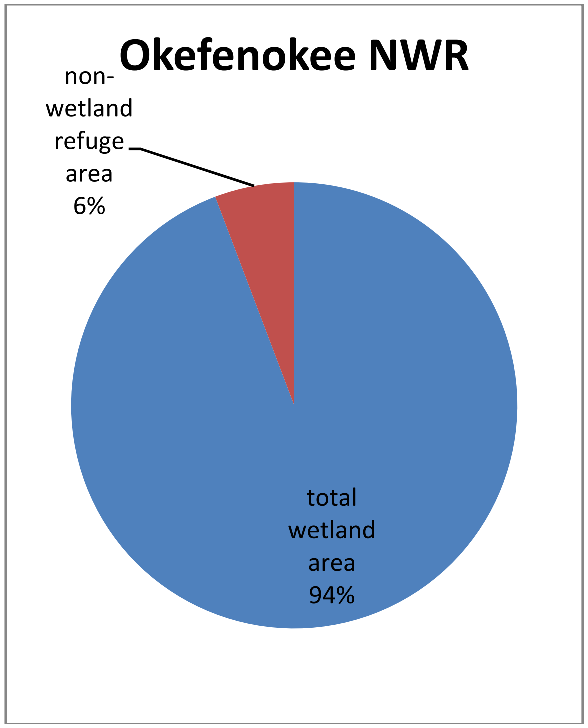


Figure 9. Proportion of Okefenokee NWR acquired acreage identified as NWI wetlands.

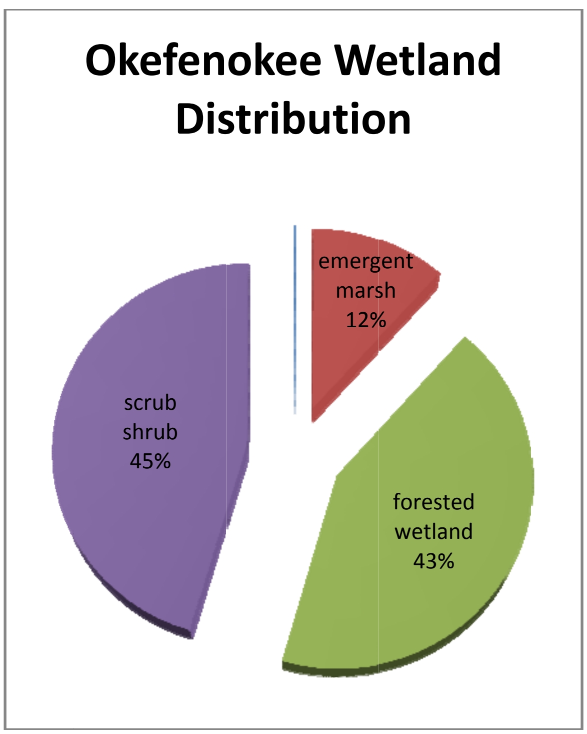


Figure 10a. Distribution of NWI wetlands at Okefenokee NWR for carbon storage analysis.

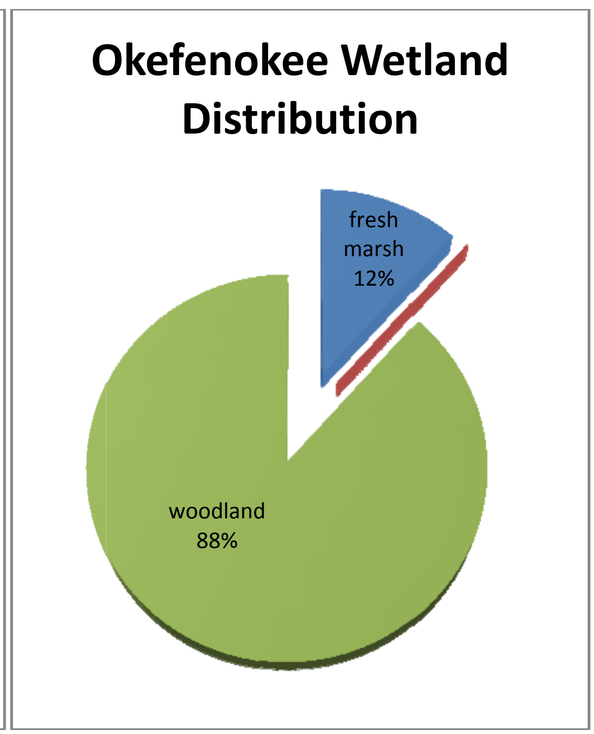


Figure 10b. Distribution of NWI wetlands at Okefenokee NWR for meta-analysis prediction.

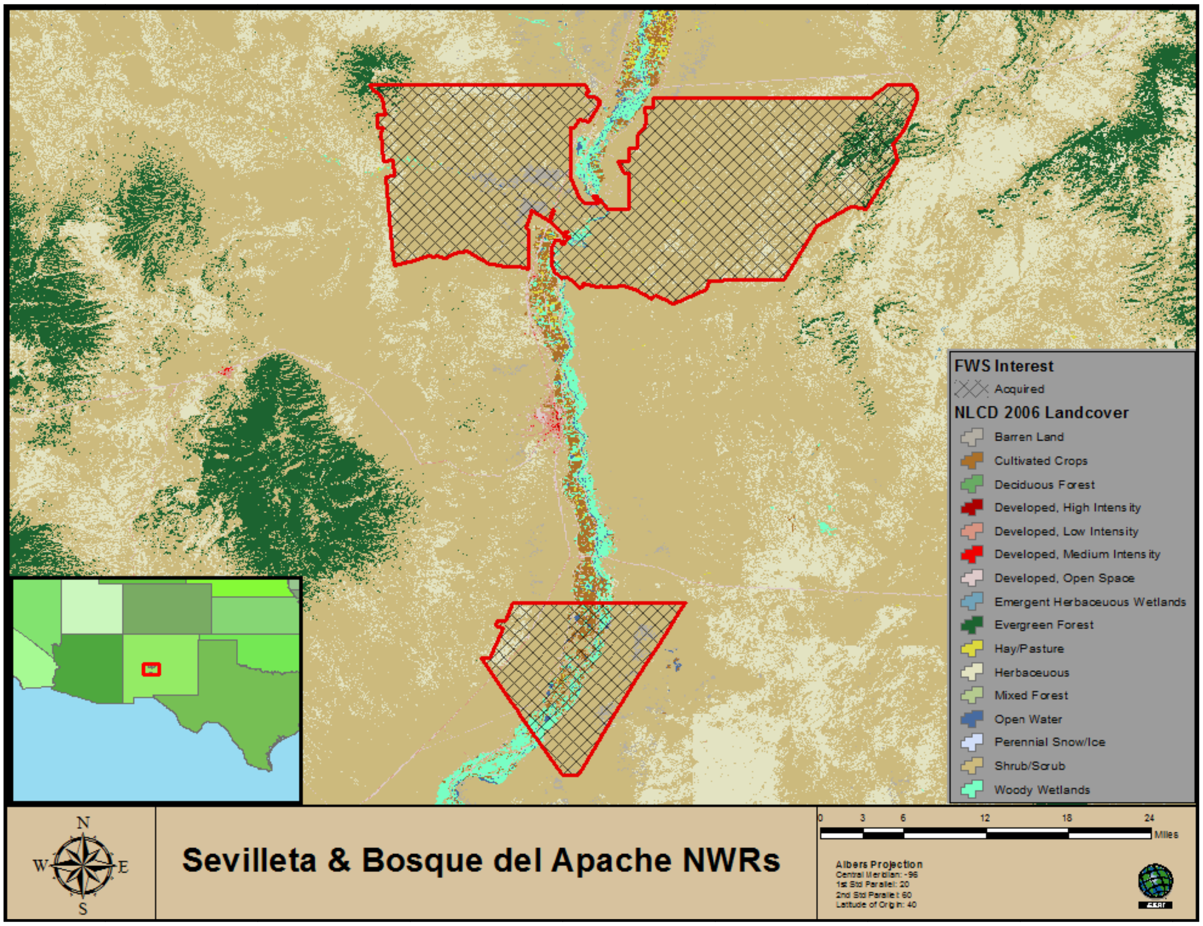


Figure 11. Sevilleta & Bosque del Apache NWRs, NLCD 2006 landcover.

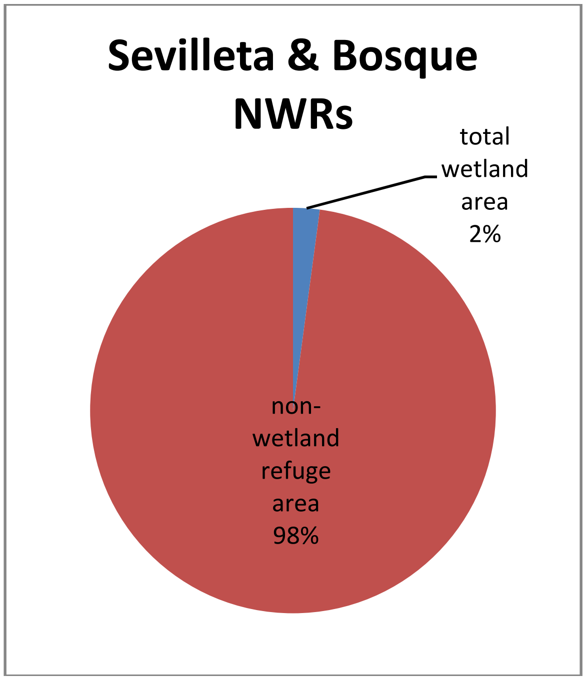


Figure 12. Proportion of Sevilleta & Bosque del Apache NWRs acquired acreage identified as NWI wetlands.

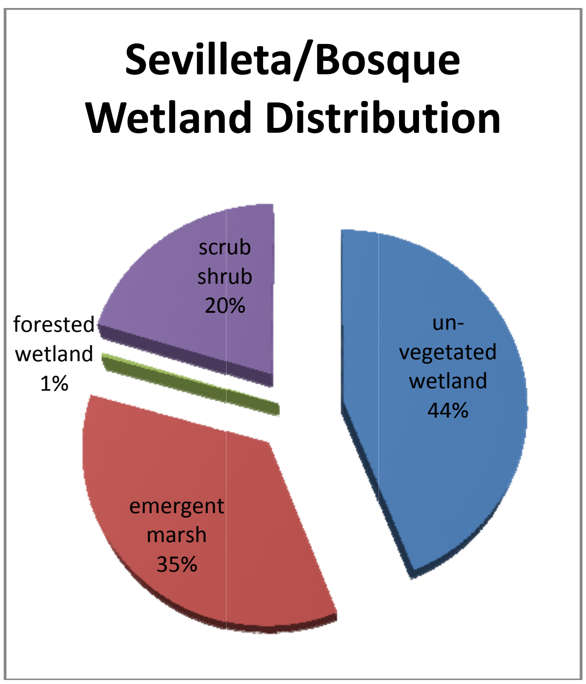


Figure 13a. Distribution of NWI wetlands at Sevilleta & Bosque del Apache NWRs for carbon storage analysis.

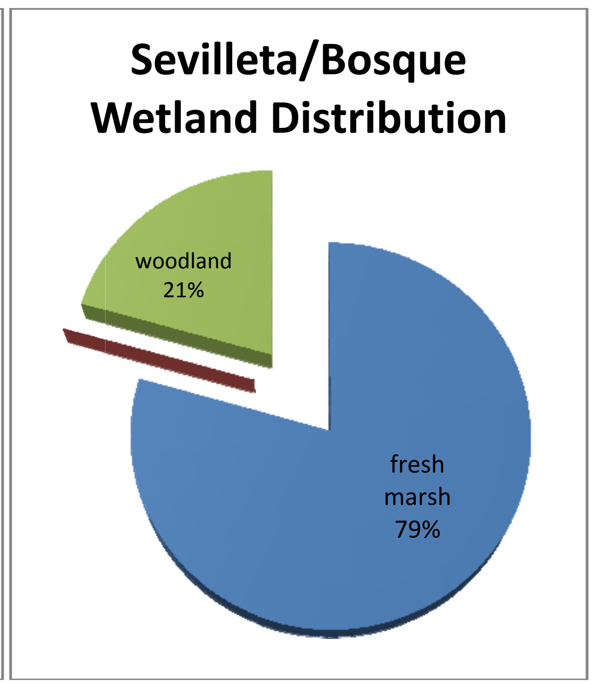


Figure 13b. Distribution of NWI wetlands at Sevilleta & Bosque del Apache NWRs for meta-analysis prediction.

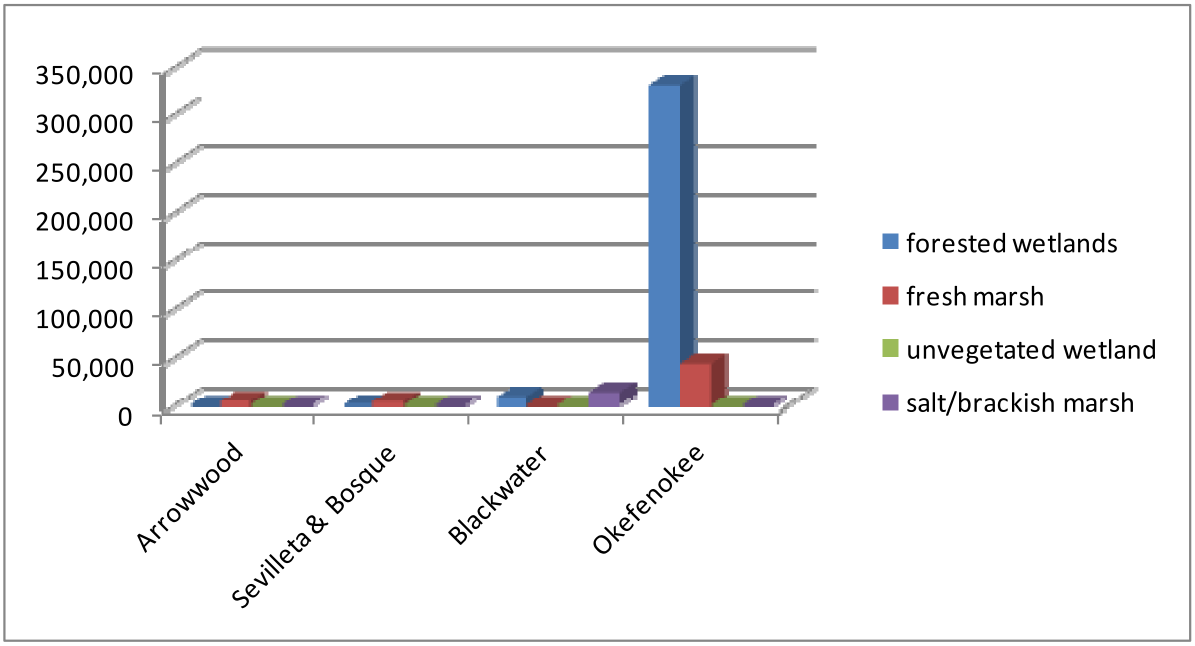


Figure 14. Acres of acquired wetlands, cross refuge comparison.

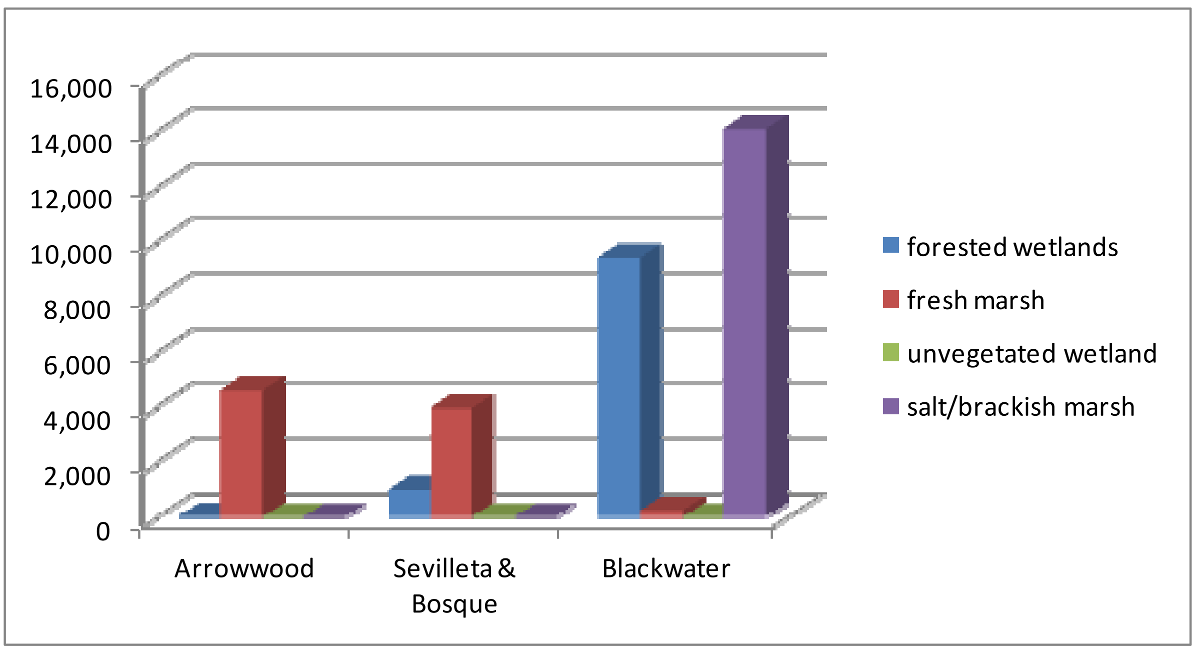


Figure 15. Acres of acquired wetlands, cross refuge comparison, detailed view of Arrowwood, Blackwater, and Sevilleta & Bosque del Apache NWRs.

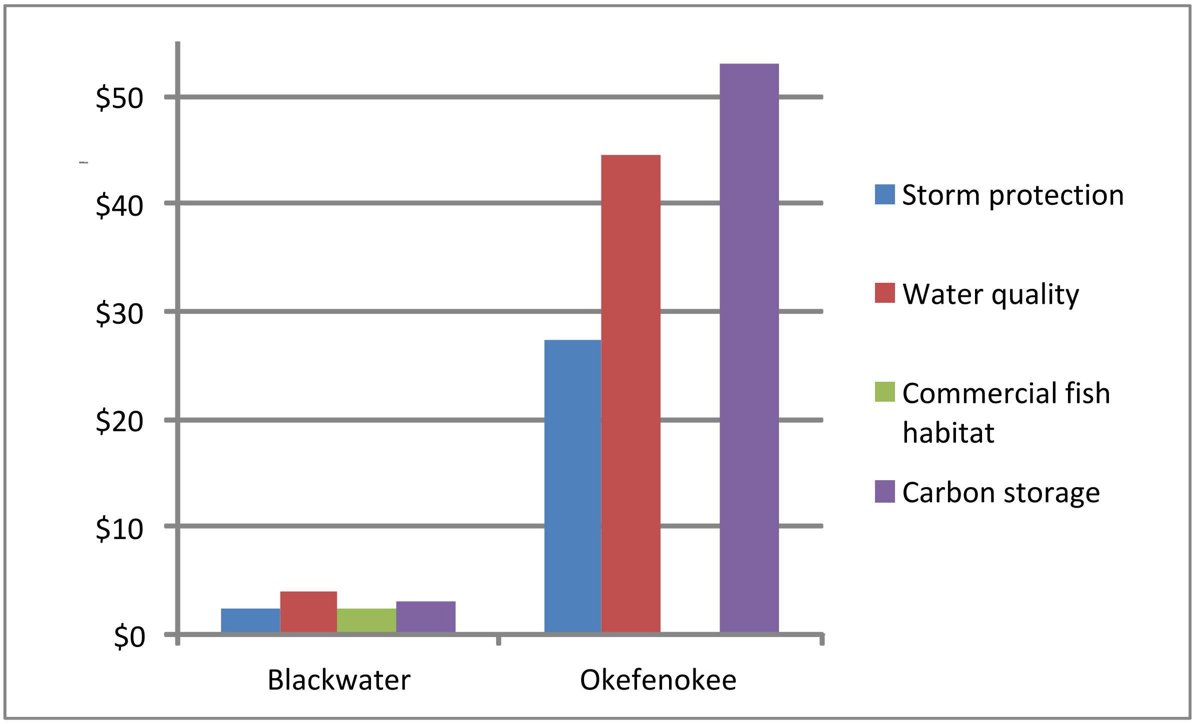


Figure 16. Comparison of aggregate annual values per year for Blackwater and Okefenokee NWRs, millions of 2010 US dollars per year.

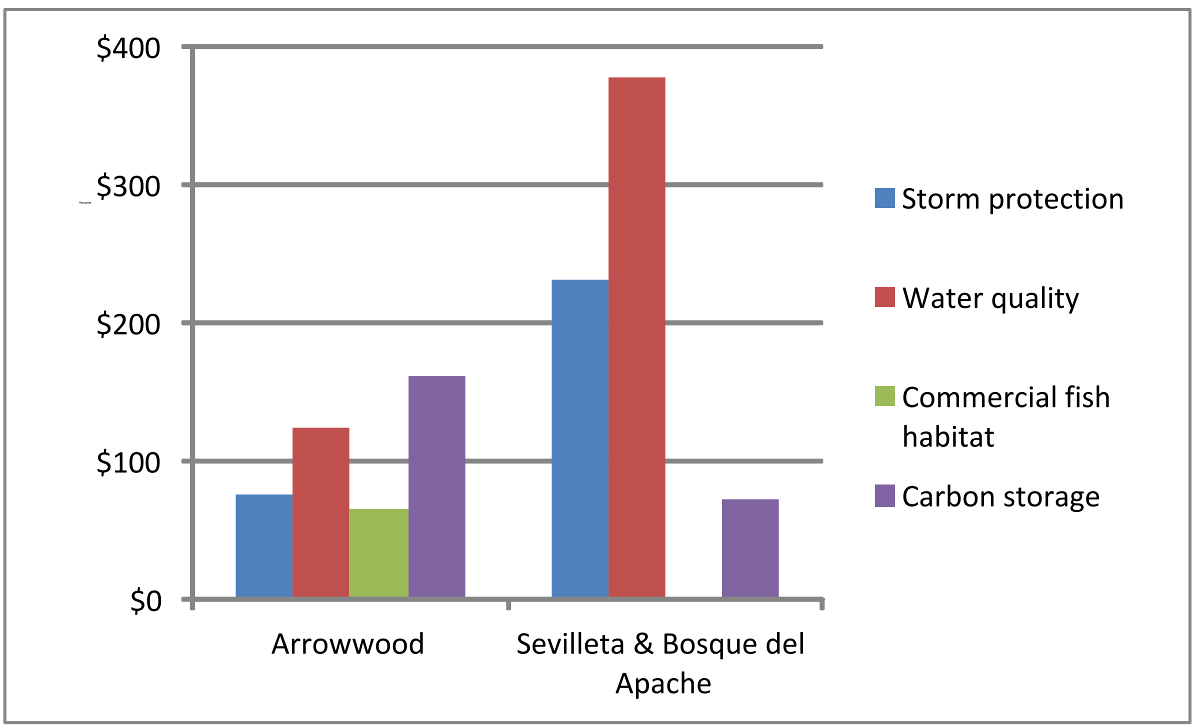


Figure 17. Comparison of aggregate annual values per year for Arrowwood and Sevilleta & Bosque del Apache NWRs

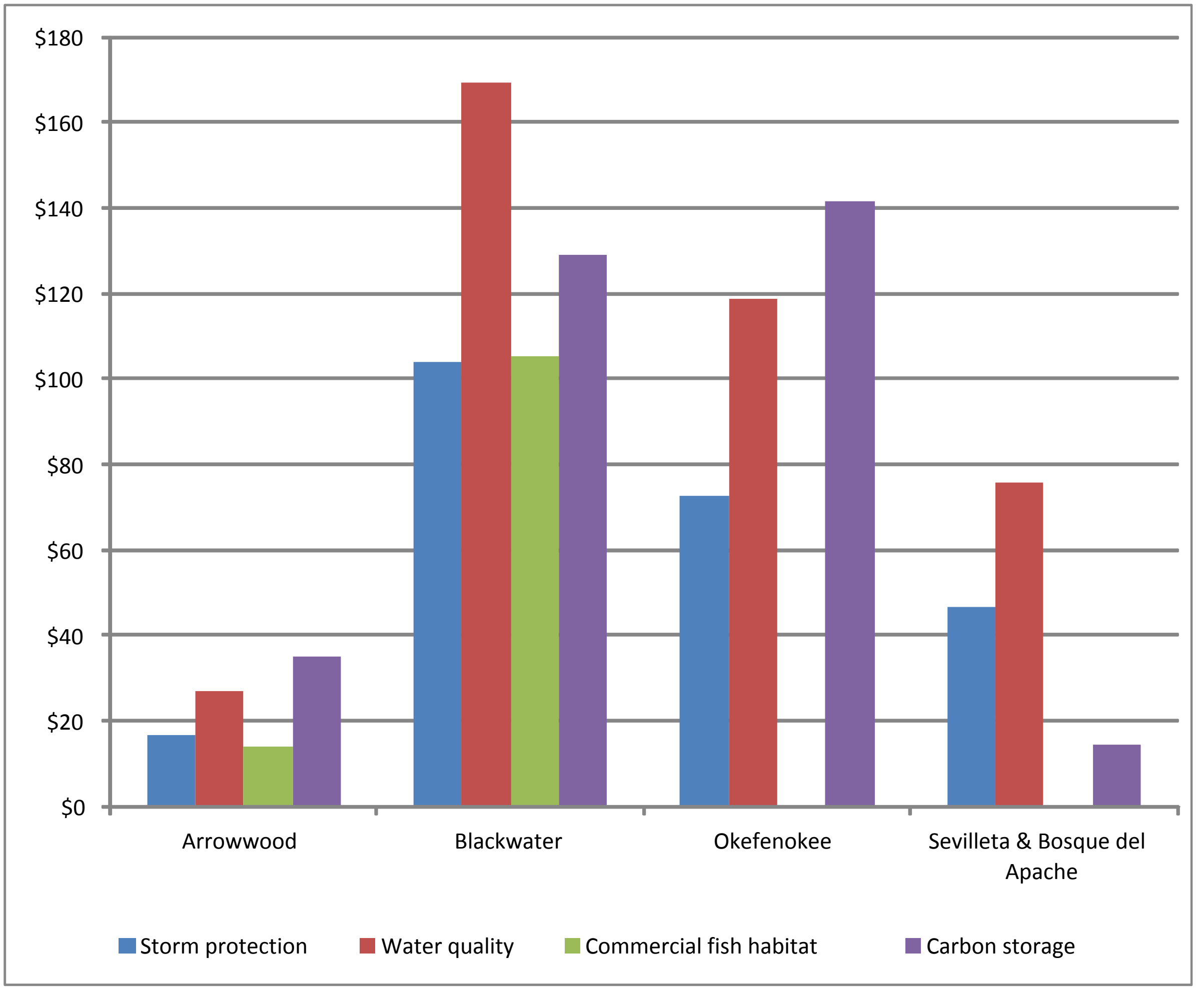


Figure 18. Cross refuge comparison of yearly value of ecosystem services for the average wetland acre, 2010 US dollars per acre per year.

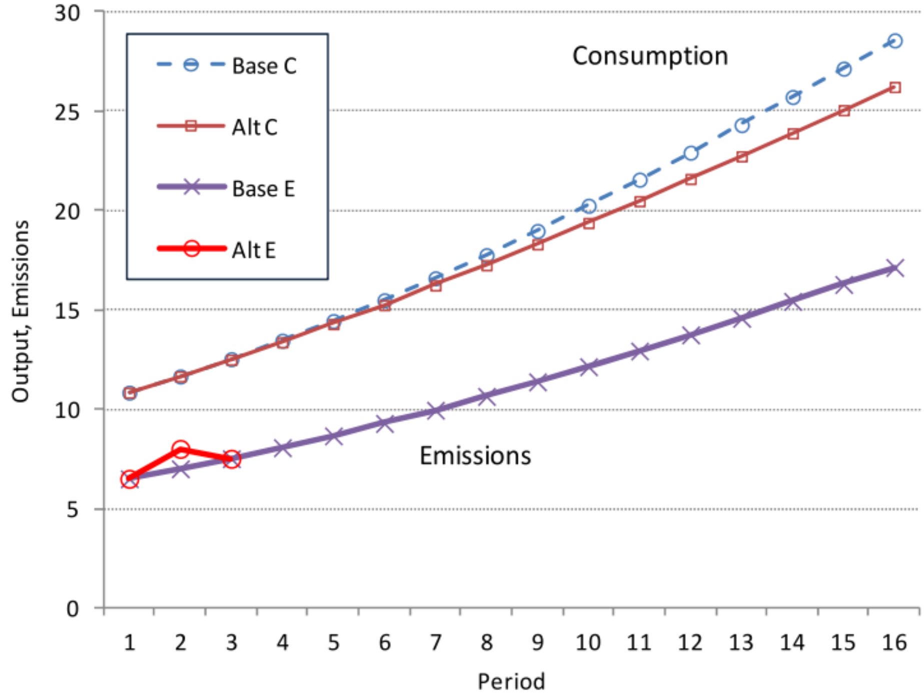


**Figure 1. Illustration of calculation of social cost of carbon**

In this example, emissions are increased by 1 unit in period 2. This leads to an alternative and lower path of economic welfare ("consumption"). The SCC is calculated as the present value of the difference in the consumption paths divided by the increment in emissions.

Figure 19. Figure 1 reproduced from Nordhaus 2011

# Tables

| Arrowwood NWR acquired wetlands | | | | |
|---|---|---|---|---|
| Qualitative value | | High | Med | Low |
| Ecosystem good or service | Commercial fish habitat | | | ☑ |
| | Water Quality provisioning | | ☑ | |
| | Carbon sequestration | | ☑ | |
| | Storm protection | ☑ | | |

Table 1. Qualitative valuation of Arrowwood NWR.

| Arrowwood NWR meta-analysis benefit transfer, value per year | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Storm protection | | Water quality provisioning | | Commercial fish habitat | | Aggregate across 3 ecosystem services | |
| | per-acre | per-refuge | per-acre | per-refuge | per-acre | per-refuge | per-acre | per-refuge |
| Contingent Valuation | $17 | $76,000 | $27 | $125,000 | $14 | $64,000 | $58 | $265,000 |

Table 2. Meta-Analysis predicted yearly value for FWS acquired wetlands of Arrowwood NWR.

| Arrowwood NWR Carbon Stocks | | | | | |
|---|---|---|---|---|---|
| total carbon (g) | $CO_2$ equivalent | | | | |
| | forested wetland | scrub-shrub wetland | emergent marsh | unvegetated wetland | all wetlands |
| above-ground living | 1.9975E+08 | 0 | 9.9616E+07 | 0 | 2.9937E+08 |
| above-ground dead | 4.0135E+07 | 0 | 0* | 0 | 4.0135E+07 |
| below-ground living | 5.1936E+07 | 0 | 9.9616E+07 | 0 | 1.5155E+08 |
| below-ground dead | 6.2913E+08 | 0 | 3.9833E+11 | 1.5549E+09 | 4.0051E+11 |
| total carbon | 9.2095E+08 | 0 | 3.9853E+11 | 1.5549E+09 | 4.0100E+11 |
| per-acre carbon (g) | $CO_2$ equivalent | | | | |
| Arrowwood | forested wetland | scrub-shrub wetland | emergent marsh | unvegetated wetland | |
| above-ground living | 2.77E+07 | 0 | 5.39E+04 | 0 | |
| above-ground dead | 5.56E+06 | 0 | 0* | 0 | |
| below-ground living | 7.19E+06 | 0 | 5.39E+04 | 0 | |
| below-ground dead | 8.72E+07 | 0 | 8.72E+07 | 8.72E+07 | |
| total carbon | 1.28E+08 | 0 | 8.73E+07 | 8.72E+07 | |
| Present value of carbon sequestration | | U.S. social cost of CO2e | | Global social cost of CO2e | |
| 1 metric ton CO2e, 2010 US dollars | | $1.10 | | $13.02 | |
| Acquired refuge wetlands | | $441,000 | | $5,200,000 | |
| Average acquired wetland acre | | $96 | | $1,100 | |
| annual value of 100 year annuitization | | $0.0338 | | $0.4002 | |
| Acquired refuge wetlands | | $14,000 | | $160,000 | |
| Average acquired wetland acre | | $3 | | $34 | |

Table 3. Estimated carbon stocks as $CO_2$equivalent by landcover and carbon pool for acquired wetlands of Arrowwood NWR.

*included in value for above-ground living carbon

| Arrowwood NWR present value (3% discount rate, 100 year horizon) | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Storm protection | | Water quality provisioning | | Commercial fish habitat | | | Carbon sequestration | | Aggregate across 4 ecosystem services | |
| | per-acre | per-refuge | per-acre | per-refuge | per-acre | per-refuge | | per-acre | per-refuge | per-acre | per-refuge |
| Contingent Valuation | $540 | $2,500,000 | $900 | $4,100,000 | $460 | $2,100,000 | Global social cost of carbon | $1,100 | $5,200,000 | $3,000 | $13,900,000 |
| | | | | | | | U.S. social cost of carbon | $100 | $440,000 | $2,000 | $9,140,000 |

Table 4. Predicted net present value for FWS acquired wetlands of Arrowwood NWR at 3% discount rate over 100 year horizon for selected services.

| Blackwater NWR acquired wetlands | | | | |
|---|---|---|---|---|
| Qualitative value | | High | Med | Low |
| Ecosystem good or service | Commercial fish habitat | | ☑ | |
| | Water Quality provisioning | ☑ | | |
| | Carbon sequestration | | ☑ | |
| | Storm protection | ☑ | | |

Table 5. Qualitative valuation of Blackwater NWR.

| Blackwater NWR meta-analysis benefit transfer, value per year | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Storm protection | | Water quality provisioning | | Commercial fish habitat | | Aggregate across 3 ecosystem services | |
| | per-acre | per-refuge | per-acre | per-refuge | per-acre | per-refuge | per-acre | per-refuge |
| Contingent Valuation | $104 | $2,500,000 | $169 | $4,000,000 | $105 | $2,500,000 | $378 | $9,000,000 |

Table 6. Meta-Analysis predicted yearly value for FWS acquired wetlands of Blackwater NWR.

| Blackwater NWR Carbon Stocks | | | | | |
|---|---|---|---|---|---|
| total carbon (g) | $CO_2$ equivalent | | | | |
| | forested wetland | scrub-shrub wetland | emergent marsh | unvegetated wetland | all wetlands |
| above-ground living | 6.5740E+11 | 2.1036E+10 | 1.6573E+10 | 0 | 6.9501E+11 |
| above-ground dead | 2.6720E+11 | 0* | 8.8716E+09 | 0 | 2.7607E+11 |
| below-ground living | 1.7092E+11 | 5.4695E+09 | 1.6573E+10 | 0 | 1.9297E+11 |
| below-ground dead | 1.3360E+12 | 5.9711E+10 | 3.0586E+12 | 2.0571E+12 | 6.5115E+12 |
| total carbon | 2.4315E+12 | 8.6217E+10 | 3.1006E+12 | 2.0571E+12 | 7.6755E+12 |
| per-acre carbon (g) | $CO_2$ equivalent | | | | |
| Blackwater | forested wetland | scrub-shrub wetland | emergent marsh | unvegetated wetland | |
| above-ground living | 7.30E+07 | 5.22E+07 | 1.93E+06 | 0 | |
| above-ground dead | 2.97E+07 | 0* | 1.03E+06 | 0 | |
| below-ground living | 1.90E+07 | 1.36E+07 | 1.93E+06 | 0 | |
| below-ground dead | 1.48E+08 | 1.48E+08 | 3.56E+08 | 3.56E+08 | |
| total carbon | 2.70E+08 | 2.14E+08 | 3.61E+08 | 3.56E+08 | |
| Present value of carbon sequestration | | U.S. social cost of CO2e | | Global social cost of CO2e | |
| 1 metric ton CO2e, 2010 US dollars | | $1.10 | | $13.02 | |
| Acquired refuge wetlands | | $8,400,000 | | $100,000,000 | |
| Average acquired wetland acre | | $350 | | $4,200 | |
| annual value of 100 year annuitization | | $0.0338 | | $0.4002 | |
| Acquired refuge wetlands | | $258,000 | | $3,072,000 | |
| Average acquired wetland acre | | $11 | | $129 | |

Table 7. Estimated carbon stocks as $CO_2$equivalent by landcover and carbon pool for acquired wetlands of Blackwater NWR.

*included in value for above-ground living carbon

| Blackwater NWR present value (3% discount rate, 100 year horizon) | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Storm protection | | Water quality provisioning | | Commercial fish habitat | | | Carbon sequestration | | Aggregate across 4 ecosystem services | |
| | per-acre | per-refuge | per-acre | per-refuge | per-acre | per-refuge | | per-acre | per-refuge | per-acre | per-refuge |
| | | | | | | | Global social cost of carbon | $4,200 | $100,000,000 | $16,200 | $392,000,000 |
| Contingent Valuation | $3,000 | $80,000,000 | $6,000 | $131,000,000 | $3,000 | $81,000,000 | | | | | |
| | | | | | | | U.S. social cost of carbon | $350 | $8,400,000 | $12,350 | $300,400,000 |

Table 8. Predicted net present value for FWS acquired wetlands of Blackwater NWR at 3% discount rate over 100 year horizon for selected services.

| Okefenokee NWR acquired wetlands | | | | |
|---|---|---|---|---|
| Qualitative value | | High | Med | Low |
| Ecosystem good or service | Commercial fish habitat | | | ☑ |
| | Water Quality provisioning | | ☑ | |
| | Carbon sequestration | ☑ | | |
| | Storm protection | | ☑ | |

Table 9. Qualitative valuation of Okefenokee NWR.

| Okefenokee NWR meta-analysis benefit transfer, value per year | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Storm protection | | Water quality provisioning | | Commercial fish habitat | | Aggregate across 3 ecosystem services | |
| | per-acre | per-refuge | per-acre | per-refuge | per-acre | per-refuge | per-acre | per-refuge |
| Contingent Valuation | $73 | $27,300,000 | $119 | $44,600,000 | $0 | $0 | $192 | $71,900,000 |

Table 10. Meta-Analysis predicted yearly value for FWS acquired wetlands of Okefenokee NWR.

| Okefenokee NWR Carbon Stocks | | | | | |
|---|---|---|---|---|---|
| total carbon (g) | $CO_2$ equivalent | | | | |
| | forested wetland | scrub-shrub wetland | emergent marsh | unvegetated wetland | all wetlands |
| above-ground living | 3.2906E+13 | 6.0009E+11 | 3.9882E+10 | 0 | 3.3546E+13 |
| above-ground dead | 3.7021E+12 | 5.9971E+10 | 2.6588E+10 | 0 | 3.7887E+12 |
| below-ground living | 8.5555E+12 | 1.5602E+11 | 1.0369E+10 | 0 | 8.7219E+12 |
| below-ground dead | 3.7071E+13 | 3.9330E+13 | 1.0001E+13 | 8.2787E+10 | 8.6485E+13 |
| total carbon | 8.2235E+13 | 4.0146E+13 | 1.0078E+13 | 8.2787E+10 | 1.3254E+14 |
| per-acre carbon (g) | $CO_2$ equivalent | | | | |
| Okefenokee | forested wetland | scrub-shrub wetland | emergent marsh | unvegetated wetland | |
| above-ground living | 2.05E+08 | 3.52E+06 | 9.19E+05 | 0 | |
| above-ground dead | 2.30E+07 | 3.51E+05 | 6.13E+05 | 0 | |
| below-ground living | 5.32E+07 | 9.14E+05 | 2.39E+05 | 0 | |
| below-ground dead | 2.30E+08 | 2.30E+08 | 2.30E+08 | 2.30E+08 | |
| total carbon | 5.11E+08 | 2.35E+08 | 2.32E+08 | 2.30E+08 | |
| Present value of carbon sequestration | | U.S. social cost of CO2e | | Global social cost of CO2e | |
| 1 metric ton CO2e, 2010 US dollars | | $1.10 | | $13.02 | |
| Acquired refuge wetlands | | $146,000,000 | | $1,726,000,000 | |
| Average acquired wetland acre | | $390 | | $4,600 | |
| annual value of 100 year annuitization | | $0.0338 | | $0.4002 | |
| Acquired refuge wetlands | | $4,486,000 | | $53,031,000 | |
| Average acquired wetland acre | | $12 | | $141 | |

Table 11. Estimated carbon stocks as $CO_2$equivalent by landcover and carbon pool for acquired wetlands of Okefenokee NWR.

| Okefenokee NWR present value (3% discount rate, 100 year horizon) | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | Storm protection | | Water quality provisioning | | Commercial fish habitat | | | Carbon sequestration | | Aggregate across 4 ecosystem services | |
| | per-acre | per-refuge | per-acre | per-refuge | per-acre | per-refuge | | per-acre | per-refuge | per-acre | per-refuge |
| | | | | | | | Global social cost of carbon | $4,600 | $1,726,000,000 | $10,500 | $4,066,000,000 |
| Contingent Valuation | $2,000 | $889,000,000 | $3,900 | $1,451,000,000 | $0 | $0 | | | | | |
| | | | | | | | U.S. social cost of carbon | $390 | $146,000,000 | $6,290 | $2,486,000,000 |

Table 12. Predicted net present value for FWS acquired wetlands of Okefenokee NWR at 3% discount rate over 100 year horizon for selected services.

| Sevilleta & Bosque del Apache NWRs acquired wetlands | | High | Med | Low |
|---|---|---|---|---|
| Qualitative value | | High | Med | Low |
| Ecosystem good or service | Commercial fish habitat | | | ☑ |
| | Water Quality provisioning | ☑ | | |
| | Carbon sequestration | | ☑ | |
| | Storm protection | | | ☑ |

Table 13. Qualitative valuation of Sevilleta & Bosque del Apache NWRs.

| Sevilleta & Bosque del Apache NWRs meta-analysis benefit transfer, value per year | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Storm protection | | Water quality provisioning | | Commercial fish habitat | | Aggregate across 3 ecosystem services | |
| | per-acre | per-refuge | per-acre | per-refuge | per-acre | per-refuge | per-acre | per-refuge |
| Contingent Valuation | $47 | $231,000 | $76 | $377,000 | $0 | $0 | $122 | $608,000 |

Table 14. Meta-Analysis predicted yearly value for FWS acquired wetlands of Sevilleta & Bosque del Apache NWRs.

| Sevilleta & Bosque del Apache NWRs Carbon Stocks | | | | | |
|---|---|---|---|---|---|
| total carbon (g) | $CO_2$ equivalent | | | | |
| | forested wetland | scrub-shrub wetland | emergent marsh | unvegetated wetland | all wetlands |
| above-ground living | 5.4096E+08 | 5.7706E+09 | 7.7666E+08 | 0 | 7.0882E+09 |
| above-ground dead | 1.0305E+08 | 0* | 0* | 0 | 1.0305E+08 |
| below-ground living | 1.4065E+08 | 4.4763E+09 | 7.7666E+08 | 0 | 5.3936E+09 |
| below-ground dead | 1.0634E+09 | 6.3280E+10 | 4.5093E+10 | 5.6180E+10 | 1.6562E+11 |
| total carbon | 1.8480E+09 | 7.3526E+10 | 4.6646E+10 | 5.6180E+10 | 1.7820E+11 |
| per-acre carbon (g) | $CO_2$ equivalent | | | | |
| Sevilleta/Bosque | forested wetland | scrub-shrub wetland | emergent marsh | unvegetated wetland | |
| above-ground living | 3.20E+07 | 5.73E+06 | 4.43E+05 | 0 | |
| above-ground dead | 6.09E+06 | 0* | 0* | 0 | |
| below-ground living | 8.32E+06 | 4.45E+06 | 4.43E+05 | 0 | |
| below-ground dead | 6.29E+07 | 6.29E+07 | 2.57E+07 | 2.57E+07 | |
| total carbon | 1.09E+08 | 7.31E+07 | 2.66E+07 | 2.57E+07 | |
| Present value of carbon sequestration | | U.S. social cost of CO2e | | Global social cost of CO2e | |
| 1 metric ton CO2e, 2010 US dollars | | $1.10 | | $13.02 | |
| Acquired refuge wetlands | | $196,000 | | $2,320,000 | |
| Average acquired wetland acre | | $40 | | $470 | |
| annual value of 100 year annuitization | | $0.0338 | | $0.4002 | |
| Acquired refuge wetlands | | $6,000 | | $71,000 | |
| Average acquired wetland acre | | $1 | | $14 | |

Table 15. Estimated carbon stocks as $CO_2$equivalent by landcover and carbon pool for acquired wetlands of Sevilleta & Bosque del Apache NWRs.

*included in value for above-ground living carbon

| Sevilleta & Bosque del Apache NWRs present value (3% discount rate, 100 year horizon) | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | Storm protection | | Water quality provisioning | | Commercial fish habitat | | | Carbon sequestration | | Aggregate across 4 ecosystem services | |
| | per-acre | per-refuge | per-acre | per-refuge | per-acre | per-refuge | | per-acre | per-refuge | per-acre | per-refuge |
| Contingent Valuation | $1,500 | $8,000,000 | $2,500 | $12,000,000 | $0 | $0 | Global social cost of carbon | $470 | $2,300,000 | $4,470 | $22,300,000 |
| | | | | | | | U.S. social cost of carbon | $40 | $200,000 | $4,040 | $20,200,000 |

Table 16. Predicted net present value for FWS acquired wetlands of Sevilleta & Bosque del Apache NWRs at 3% discount rate over 100 year horizon for selected services.

| Inter-study comparison of results | | | | | |
|---|---|---|---|---|---|
| | | Arrowwood NWR | Blackwater NWR | Okefenokee NWR | Sevilleta & Bosque del Apache NWRs |
| | NPP group # | 3 | 5 | 7 | 1 |
| | **Ingraham and Foster's method** | | | | |
| per acre values | Disturbance prevention | $1,280 | $1,689 | $2,236 | $644 |
| | Freshwater regulation and supply | $1,392 | $1,840 | $2,437 | $697 |
| | Habitat provision | $136 | $171 | $216.28* | $83* |
| | | | | | |
| aggregate values | FWS acquired wetland acres | 4,595 | 23,788 | 375,261 | 4,958 |
| | Estimated value following Ingraham and Foster (2008) | $13,000,000 | $88,000,000 | $1,753,000,000 | $6,600,000 |
| | | | | | |
| | Final valuation results (CVM) | $265,357 | $8,999,971 | $71,889,402 | $607,280 |
| | ratio of our results to Ingraham and Foster's results | 2% | 10% | 4% | 9% |

Table 17. Comparison of our results with results derived from Ingraham and Foster (2008). Annual ecosystem service flows, 2010 US dollars.

* habitat value set to zero for consistency with our results

| **Valuation Linkages** | **Examples** |
|---|---|
| Ecosystem Structure | Wetland Vegetation, Water Quantity and Quality, Nutrient Levels, Fish and Wildlife Populations |
| 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 |
| Ecosystem Values | WTP for Fishing or Hunting Day<br>WTP for Wildlife Observation Day<br>WTP for Carbon Sequestration<br>Flood Damage Avoidance |

Table 18. Ecosystem service conceptual linkages.

| Variable | Coefficient (β) | standard error | Arrowwood NWR regressors (X) | Arrowwood NWR Xβ | Blackwater NWR regressors (X) | Blackwater NWR Xβ | Okefenokee NWR regressors (X) | Okefenokee NWR Xβ | Sevilleta & Bosque del Apache NWRs regressors (X) | Sevilleta & Bosque del Apache NWRs Xβ |
|---|---|---|---|---|---|---|---|---|---|---|
| Intercept | -6.98 | 4.67 | 1 | -6.98 | 1 | -6.98 | 1 | -6.98 | 1 | -6.98 |
| Structure and function variables | | | | | | | | | | |
| ln(hectares of wetland) | -0.11 | 0.05 | 7.528143345 | -0.8280958 | 9.172312169 | -1.0089543 | 11.93073155 | -1.3123805 | 7.604019746 | -0.8364422 |
| Absolute val of latitude | 0.03 | 0.07 | 47.2395 | 1.417185 | 38.4 | 1.152 | 30.837014 | 0.92511042 | 34.0535 | 1.021605 |
| Latitude squared | -0.0007 | 0.001 | 2231.57036 | -1.5620993 | 1474.56 | -1.032192 | 950.9214324 | -0.665645 | 1159.640862 | -0.8117486 |
| Mangrove | -0.56 | 0.82 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Unvegetated sediment | 0.22 | 1.09 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Salt-brackish marsh | -0.31 | 0.42 | 0 | 0 | 0.593451704 | -0.18397 | 0 | 0 | 0 | 0 |
| Fresh marsh | -1.46 | 0.59 | 0.998429197 | -1.4577066 | 0.010853901 | -0.0158467 | 0.116594269 | -0.1702276 | 0.793607792 | -1.1586674 |
| Woodland | 0.86 | 0.42 | 0.001570803 | 0.00135089 | 0.395694395 | 0.34029718 | 0.883405731 | 0.75972893 | 0.206392208 | 0.1774973 |
| User population variables | | | | | | | | | | |
| ln(GDP per capita) | 1.16 | 0.46 | 10.40759063 | 12.0728051 | 10.40759063 | 12.0728051 | 10.40759063 | 12.0728051 | 10.40759063 | 12.0728051 |
| ln(pop density) | 0.47 | 0.12 | -5.841684289 | -2.7455916 | -2.711530646 | -1.2744194 | -4.073003218 | -1.9143115 | -5.398196299 | -2.5371523 |
| South America | 0.23 | 1.19 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Europe | 0.84 | 0.92 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Asia | 2.01 | 1.34 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Africa | 3.51 | 1.52 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Australasia | 1.75 | 0.94 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Urban | 1.11 | 0.48 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Study variables | | | | | | | | | | |
| CVM | 1.49 | 0.73 | 1 | 1.49 | 1 | 1.49 | 1 | 1.49 | 1 | 1.49 |
| Hedonic pricing | -0.71 | 1.54 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| TCM | 0.01 | 0.65 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Replacement cost | 0.63 | 0.81 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Net factor income | 0.19 | 0.61 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Production function | -1 | 0.75 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Market prices | -0.04 | 0.53 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Opportunity cost | -0.03 | 0.72 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Marginal | 0.95 | 0.48 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Flood control | 0.14 | 0.55 | 1 | 0.14 | 1 | 0.14 | 1 | 0.14 | 1 | 0.14 |
| Water supply | -0.95 | 0.71 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Water quality | 0.63 | 0.74 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Habitat and nursery | -0.03 | 0.35 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Hunting | -1.1 | 0.43 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Fishing | 0.06 | 0.36 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Material | -0.83 | 0.42 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Fuelwood | -1.24 | 0.45 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Amenity | 0.06 | 0.39 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Biodiversity | 0.06 | 0.81 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Ramsar proportion | -1.32 | 0.7 | 0 | 0 | 1 | -1.32 | 1 | -1.32 | 0 | 0 |
| | | | column sum of Xβ | 1.54784776 | | 3.37971985 | | 3.02507986 | | 2.57789702 |
| 1995 US dollars | per hectare | | exp(Xβ) | $ 4.70 | | $ 29.36 | | $ 20.60 | | $ 13.17 |
| | per acre | | exp(Xβ)*2.471 | $ 11.62 | | $ 72.56 | | $ 50.89 | | $ 32.54 |
| | 2010 US dollars per acre | | 1995 dollars x 1.43 | $ 16.61 | | $ 103.76 | | $ 72.78 | | $ 46.54 |

Table 19, Meta-analysis benefit transfer for flood control valued by the Contingent Valuation Method. Bold values under the “Study variables” category must be 0 or 1 and sum to exactly 1 in each of 8 boxes.

THE EMPIRICS OF WETLAND VALUATION

*Table III.* Meta-regression results[a]

| Category | Variable[b] | Coefficient | Standard error |
|---|---|---|---|
| | Constant | −6.98 | 4.67 |
| Socio-economic | GDP per capita (log) | 1.16** | 0.46 |
| | Population density (log) | 0.47*** | 0.12 |
| Geographic characteristics | Wetland size (log) | −0.11** | 0.05 |
| | Latitude (absolute value) | 0.03 | 0.07 |
| | Latitude squared | −0.0007 | 0.0010 |
| | South America | 0.23 | 1.19 |
| | Europe | 0.84 | 0.92 |
| | Asia | 2.01 | 1.34 |
| | Africa | 3.51** | 1.52 |
| | Australasia | 1.75* | 0.94 |
| | Urban | 1.11** | 0.48 |
| Valuation methods | CVM | 1.49** | 0.73 |
| | Hedonic pricing | −0.71 | 1.54 |
| | TCM | 0.01 | 0.65 |
| | Replacement cost | 0.63 | 0.81 |
| | Net factor income | 0.19 | 0.61 |
| | Production function | −1.00 | 0.75 |
| | Market prices | −0.04 | 0.53 |
| | Opportunity cost | −0.03 | 0.72 |
| Type value | Marginal | 0.95* | 0.48 |
| Wetland type | Mangrove | −0.56 | 0.82 |
| | Unvegetated sediment | 0.22 | 1.09 |
| | Salt/brackish marsh | −0.31 | 0.42 |
| | Fresh marsh | −1.46** | 0.59 |
| | Woodland | 0.86** | 0.42 |
| Wetland service | Flood control | 0.14 | 0.55 |
| | Water supply | −0.95 | 0.71 |
| | Water quality | 0.63 | 0.74 |
| | Habitat and nursery | −0.03 | 0.35 |
| | Hunting | −1.10** | 0.43 |
| | Fishing | 0.06 | 0.36 |
| | Material | −0.83** | 0.42 |
| | Fuelwood | −1.24*** | 0.45 |
| | Amenity | 0.06 | 0.39 |
| | Biodiversity | 0.06 | 0.81 |
| RAMSAR | RAMSAR proportion | −1.32* | 0.70 |
| | *n* | 202 | |

Table 20. Estimated coefficients and standard errors, reproduced from Brander et al. 2006.

| | | NWI wetland attribute |
|---|---|---|
| Meta-analysis wetland classification | Fresh marsh | L1UBH, L1UBHh, L2AB3G, L2AB3H, L2ABGH, L2UBFh, L2UBKFh, L2USCh, L2USKCh, PAB/EMF, PAB/EMFH, PAB3/EM1G, PAB3/SS3F, PAB3G, PAB3Gh, PAB3H, PAB3Hh, PAB3Hx, PAB4Hx, PABFh, PABFx, PEM/ABF, PEM/ABFH, PEM/FOA, PEM/FOC, PEM/SS1F, PEM1/AB3B, PEM1/AB3G, PEM1/FO1A, PEM1/FO1C, PEM1/FO2C, PEM1/SS1A, PEM1/SS1C, PEM1/SS1Ch, PEM1/SS1KAh, PEM1/SS1KFh, PEM1/SS2F, PEM1/SS3B, PEM1/SS3C, PEM1/SS3F, PEM1/SS3G, PEM1/SS4F, PEM1/UBF, PEM1A, PEM1Ah, PEM1B, PEM1C, PEM1Ch, PEM1E, PEM1F, PEM1Fd, PEM1Fh, PEM1Fx, PEM1G, PEM1Gh, PEM1Jh, PEM1KAh, PEM1KCh, PEM1KFh, PEM1R, PEM1Rh, PEMA, PEMAh, PEMAx, PEMC, PEMCh, PEMF, PEMFh, PFO1/EM1A, PFO1/EM1G, PFO2/EM1C, PFO2/EM1F, PFO4/EM1C, PFO4/EM1Ch, PSS/EM1F, PSS1/EM1Ch, PSS1/EM1F, PSS1/EM1Fh, PSS1/EM1Gh, PSS2/EM1C, PSS2/EM1F, PSS3/EM1B, PSS3/EM1C, PSS3/EM1F, PSS4/EM1C, PSS5/EM1Cd, PUB, PUB/EM1Fx, PUB/SS1Fh, PUBFh, PUBFx, PUBGh, PUBH, PUBHh, PUBHx, PUBKFh, PUBKHh, PUBVh, PUS, PUSA, PUSCh, PUSJh, PUSKAh, PUSKCh, R2UBF, R2UBH, R2UBHx, R2USA, R2USC, R4SB, R4USA, R4USF |
| | Salt-brackish marsh | E1UBL, E1UBL6, E1UBLx, E2EM1/FO5P, E2EM1/SS1P, E2EM1/SS1P6, E2EM1N, E2EM1P, E2EM1P6, E2EM1U, E2EM1U6, E2FO4/EM1P, E2FO5/EM1P, E2SS/EM1P, E2SS4/EM1P, E2US3M, E2USM, E2USN |
| | Woodland wetland | E2FO1/4P, E2FO1/SS1P, E2FO1P, E2FO4/1P, E2FO4/5P, E2FO4P, E2FO4P6, E2FO5/1P, E2FO5/4P, E2FO5P, E2SS1P, E2SS1P6, PFO1/2C, PFO1/2F, PFO1/2Fh, PFO1/3B, PFO1/3Bh, PFO1/3C, PFO1/3F, PFO1/4A, PFO1/4B, PFO1/4C, PFO1/4Cd, PFO1/4E, PFO1/4F, PFO1/4R, PFO1/4S, PFO1/SS1A, PFO1/SS1Ah, PFO1/SS1Ax, PFO1/SS1KAh, PFO1/SS1KCh, PFO1/SS2A, PFO1/SS3Bh, PFO1/SS4A, PFO1A, PFO1B, PFO1C, PFO1Cd, PFO1Ch, PFO1E, PFO1Eh, PFO1F, PFO1Fh, PFO1KAh, PFO1R, PFO1S, PFO2/1B, PFO2/1F, PFO2/3F, PFO2/4A, PFO2/4B, PFO2/4C, PFO2/SS1F, PFO2/SS3B, PFO2A, PFO2B, PFO2C, PFO2F, PFO2Fh, PFO3/1B, PFO3/1C, PFO3/4B, PFO3/4C, PFO3/4F, PFO3/6C, PFO3/6F, PFO3/SS3B, PFO3B, PFO3F, PFO3Fh, PFO4/1A, PFO4/1Ah, PFO4/1B, PFO4/1C, |

| | | |
|---|---|---|
| | | PFO4/2B, PFO4/2C, PFO4/3B, PFO4/3C, PFO4/5V, PFO4/SS3B, PFO4/SS3F, PFO4A, PFO4Ad, PFO4Ah, PFO4B, PFO4C, PFO4E, PFO4F, PFO4S, PFO5Rh, PFO5V, PFO6/3C, PFO6/3F, PFO6/4C, PFO6/4F, PFO6/SS3B, PFO6/SS3F, PFO6/SS6F, PFO6B, PFO6C, PFO6F, PFO7B, PFO7C, PFOAH, PFOC, PSS, PSS1/2A, PSS1/2Ah, PSS1/2Ax, PSS1/2B, PSS1/2C, PSS1/2F, PSS1/2Fh, PSS1/2G, PSS1/2Gh, PSS1/2KAh, PSS1/3A, PSS1/3B, PSS1/3Bd, PSS1/3C, PSS1/3F, PSS1/3Fh, PSS1/3G, PSS1/4C, PSS1/4F, PSS1/FO1A, PSS1/FO1C, PSS1/FO4C, PSS1A, PSS1Ah, PSS1Ax, PSS1B, PSS1C, PSS1Ch, PSS1Cx, PSS1Eh, PSS1F, PSS1Gh, PSS1KAh, PSS1KCh, PSS1R, PSS2/1B, PSS2/1F, PSS2/1Gh, PSS2/3B, PSS2/4C, PSS2A, PSS2Ah, PSS2Ax, PSS2B, PSS2C, PSS2F, PSS2G, PSS2KAh, PSS3/1B, PSS3/1C, PSS3/1F, PSS3/2C, PSS3/2F, PSS3/4B, PSS3/4C, PSS3/4F, PSS3/6C, PSS3/6F, PSS3/FO4B, PSS3A, PSS3B, PSS3C, PSS3F, PSS4/1C, PSS4A, PSS4B, PSS4C, PSS4Fx, PSS5KFh, PSS6/3C, PSS6/3F, PSS6B, PSS6C, PSS6F, PSS7B, PSS7C |
| Carbon storage wetland classification | Emergent marsh | L2AB3G, L2AB3H, L2ABGH, PAB/EMF, PAB/EMFH, PAB3/EM1G, PAB3/SS3F, PAB3G, PAB3Gh, PAB3H, PAB3Hh, PAB3Hx, PAB4Hx, PABFh, PABFx, PEM/ABF, PEM/ABFH, PEM/FOA, PEM/FOC, PEM/SS1F, PEM1/AB3B, PEM1/AB3G, PEM1/FO1A, PEM1/FO1C, PEM1/FO2C, PEM1/SS1A, PEM1/SS1C, PEM1/SS1Ch, PEM1/SS1KAh, PEM1/SS1KFh, PEM1/SS2F, PEM1/SS3B, PEM1/SS3C, PEM1/SS3F, PEM1/SS3G, PEM1/SS4F, PEM1/UBF, PEM1A, PEM1Ah, PEM1B, PEM1C, PEM1Ch, PEM1E, PEM1F, PEM1Fd, PEM1Fh, PEM1Fx, PEM1G, PEM1Gh, PEM1Jh, PEM1KAh, PEM1KCh, PEM1KFh, PEM1R, PEM1Rh, PEMA, PEMAh, PEMAx, PEMC, PEMCh, PEMF, PEMFh, PFO1/EM1A, PFO1/EM1G, PFO2/EM1C, PFO2/EM1F, PFO4/EM1C, PFO4/EM1Ch, PSS/EM1F, PSS1/EM1Ch, PSS1/EM1F, PSS1/EM1Fh, PSS1/EM1Gh, PSS2/EM1C, PSS2/EM1F, PSS3/EM1B, PSS3/EM1C, PSS3/EM1F, PSS4/EM1C, PSS5/EM1Cd, E2EM1/FO5P, E2EM1/SS1P, E2EM1/SS1P6, E2EM1N, E2EM1P, E2EM1P6, E2EM1U, E2EM1U6, E2FO4/EM1P, E2FO5/EM1P, E2SS/EM1P, E2SS4/EM1P |
| | Forested wetland | E2FO1/4P, E2FO1P, E2FO4/1P, E2FO4/5P, E2FO4P, E2FO4P6, E2FO5/1P, E2FO5/4P, E2FO5P, PFO1/2C, PFO1/2F, PFO1/2Fh, PFO1/3B, PFO1/3Bh, PFO1/3C, PFO1/3F, PFO1/4A, PFO1/4B, PFO1/4C, PFO1/4Cd, |

| | | |
|---|---|---|
| | | PFO1/4E, PFO1/4F, PFO1/4R, PFO1/4S, PFO1A, PFO1B, PFO1C, PFO1Cd, PFO1Ch, PFO1E, PFO1Eh, PFO1F, PFO1Fh, PFO1KAh, PFO1R, PFO1S, PFO2/1B, PFO2/1F, PFO2/3F, PFO2/4A, PFO2/4B, PFO2/4C, PFO2A, PFO2B, PFO2C, PFO2F, PFO2Fh, PFO3/1B, PFO3/1C, PFO3/4B, PFO3/4C, PFO3/4F, PFO3/6C, PFO3/6F, PFO3B, PFO3F, PFO3Fh, PFO4/1A, PFO4/1Ah, PFO4/1B, PFO4/1C, PFO4/2B, PFO4/2C, PFO4/3B, PFO4/3C, PFO4/5V, PFO4A, PFO4Ad, PFO4Ah, PFO4B, PFO4C, PFO4E, PFO4F, PFO4S, PFO5Rh, PFO5V, PFO6/3C, PFO6/3F, PFO6/4C, PFO6/4F, PFO6B, PFO6C, PFO6F, PFO7B, PFO7C, PFOAH, PFOC |
| | Unvegetated wetland | L1UBH, L1UBHh, L2UBFh, L2UBKFh, L2USCh, L2USKCh, PUB, PUB/EM1Fx, PUB/SS1Fh, PUBFh, PUBFx, PUBGh, PUBH, PUBHh, PUBHx, PUBKFh, PUBKHh, PUBVh, PUS, PUSA, PUSCh, PUSJh, PUSKAh, PUSKCh, R2UBF, R2UBH, R2UBHx, R2USA, R2USC, R4SB, R4USA, R4USF, E1UBL, E1UBL6, E1UBLx, E2US3M, E2USM, E2USN |
| | Scrub-shrub wetland | E2FO1/SS1P, E2SS1P, E2SS1P6, PFO1/SS1A, PFO1/SS1Ah, PFO1/SS1Ax, PFO1/SS1KAh, PFO1/SS1KCh, PFO1/SS2A, PFO1/SS3Bh, PFO1/SS4A, PFO2/SS1F, PFO2/SS3B, PFO3/SS3B, PFO4/SS3B, PFO4/SS3F, PFO6/SS3B, PFO6/SS3F, PFO6/SS6F, PSS, PSS1/2A, PSS1/2Ah, PSS1/2Ax, PSS1/2B, PSS1/2C, PSS1/2F, PSS1/2Fh, PSS1/2G, PSS1/2Gh, PSS1/2KAh, PSS1/3A, PSS1/3B, PSS1/3Bd, PSS1/3C, PSS1/3F, PSS1/3Fh, PSS1/3G, PSS1/4C, PSS1/4F, PSS1/FO1A, PSS1/FO1C, PSS1/FO4C, PSS1A, PSS1Ah, PSS1Ax, PSS1B, PSS1C, PSS1Ch, PSS1Cx, PSS1Eh, PSS1F, PSS1Gh, PSS1KAh, PSS1KCh, PSS1R, PSS2/1B, PSS2/1F, PSS2/1Gh, PSS2/3B, PSS2/4C, PSS2A, PSS2Ah, PSS2Ax, PSS2B, PSS2C, PSS2F, PSS2G, PSS2KAh, PSS3/1B, PSS3/1C, PSS3/1F, PSS3/2C, PSS3/2F, PSS3/4B, PSS3/4C, PSS3/4F, PSS3/6C, PSS3/6F, PSS3/FO4B, PSS3A, PSS3B, PSS3C, PSS3F, PSS4/1C, PSS4A, PSS4B, PSS4C, PSS4Fx, PSS5KFh, PSS6/3C, PSS6/3F, PSS6B, PSS6C, PSS6F, PSS7B, PSS7C |

Table 21. Attribute mapping of wetland types from NWI subclasses to meta-analysis and carbon storage modeling categories.

| **Arrowwood NWR Wetland Distribution** (m²) | | | |
|---|---|---|---|
| Carbon model | | Meta-analysis model | |
| unvegetated wetland | 72,197 | fresh marsh | 18,567,284 |
| emergent marsh | 18,495,087 | | |
| forested wetland | 29,211 | woodland | 29,211 |
| scrub shrub | 0 | | |
| total | 18,596,496 | | 18,596,496 |

| **Blackwater NWR Wetland Distribution** (m²) | | | |
|---|---|---|---|
| Carbon model | | Meta-analysis model | |
| unvegetated wetland | 23,393,233 | fresh marsh | 1,044,890 |
| emergent marsh | 34,782,409 | | |
| forested wetland | 36,463,271 | woodland | 38,092,937 |
| scrub shrub | 1,629,666 | | |
| | | salt-brackish marsh | 57,130,752 |
| total | 96,268,579 | | 96,268,579 |

| **Okefenokee NWR Wetland Distribution** (m²) | | | |
|---|---|---|---|
| Carbon model | | Meta-analysis model | |
| unvegetated wetland | 1,453,684 | fresh marsh | 177,063,108 |
| emergent marsh | 175,609,424 | | |
| forested wetland | 650,947,373 | woodland | 1,341,563,058 |
| scrub shrub | 690,615,685 | | |
| total | 1,518,626,166 | | 1,518,626,166 |

| **Sevilleta & Bosque del Apache NWRs Wetland Distribution** (m²) | | | |
|---|---|---|---|
| Carbon model | | Meta-analysis model | |
| unvegetated wetland | 8,832,407 | fresh marsh | 15,921,711 |
| emergent marsh | 7,089,304 | | |
| forested wetland | 68,432 | woodland | 4,140,732 |
| scrub shrub | 4,072,300 | | |
| total | 20,062,443 | | 20,062,443 |

Table 22. Estimated wetland surface areas based on National Wetlands Inventory dataset and FWS Cadastral. Units in square meters must be converted to the natural log of hectares for the meta-analysis benefit transfer wetland area category. The ratios of fresh marsh to total wetlands and woodland marsh to total wetlands are used for the explanatory variables fresh marsh and woodland.

| | County | Land area (km$^2$) | Population |
|---|---|---|---|
| **Arrowwood NWR** | Eddy | 1632.134239 | 2388 |
| | Foster | 1645.814578 | 3447 |
| | Stutsman | 5754.239519 | 20394 |
| | column sum | 9032.188336 | 26229 |
| | Population per km$^2$ | | 2.9039474 |

| | County | Land area (km$^2$) | Population |
|---|---|---|---|
| **Blackwater NWR** | Caroline | 827.292416 | 33138 |
| | Talbot | 695.511159 | 36215 |
| | Somerset | 828.079452 | 26119 |
| | Wicomico | 969.804281 | 94046 |
| | Dorchester | 1400.575316 | 31998 |
| | St. Mary's | 925.092041 | 101578 |
| | Calvert | 552.061078 | 88698 |
| | column sum | 6198.415743 | 411792 |
| | Population per km$^2$ | | 66.43504 |

| | County | Land area (km$^2$) | Population |
|---|---|---|---|
| **Okefenokee NWR** | Ware | 2311.462171 | 35879 |
| | Brantley | 1145.713216 | 15511 |
| | Echols | 1074.571687 | 4063 |
| | Clinch | 2072.566618 | 7060 |
| | Charlton | 2003.553604 | 10848 |
| | Pierce | 819.714945 | 18127 |
| | Atkinson | 878.994487 | 8181 |
| | Columbia | 2065.707648 | 110627 |
| | Baker | 1515.7412 | 26164 |
| | column sum | 13888.02558 | 236460 |
| | Population per km$^2$ | | 17.026178 |

| | County | Land area (km$^2$) | Population |
|---|---|---|---|
| **Sevilleta & Bosque del Apache NWRs** | Valencia | 2761.38035 | 72207 |
| | Socorro | 17214.81778 | 18180 |
| | column sum | 19976.19813 | 90387 |
| | Population per km$^2$ | | 4.5247349 |

Table 23, Local population data for approximately a 50km radius around wetlands in each refuge. Values used for meta-analysis benefit transfer are the natural log of 1000 people per km$^2$.

| | 2015 | 2025 | 2035 | 2045 | 2055 |
|---|---|---|---|---|---|
| Base parameters | | | | | |
| No controls | 42.68 | 69.68 | 88.58 | 121.10 | 161.06 |
| Optimal controls | 40.11 | 65.32 | 82.51 | 111.91 | 147.41 |
| Low discount run | 138.21 | 221.43 | 246.01 | 333.95 | 442.91 |
| Stern Review | 288.35 | 364.21 | 487.67 | 627.47 | 759.01 |
| 2 degree damage | | | | | |
| Average | 97.87 | 160.07 | 203.57 | 278.50 | 369.48 |
| Maximum | 124.86 | 202.84 | 254.48 | 343.97 | 450.42 |

**Table 2. Global social cost of carbon by different assumptions**

The social cost of carbon is measured in 2005 international US dollars. Countries' GDP are calculated using purchasing power parity exchange rates. To calculate the SCC per unit of $CO_2$, the figures should be divided by 3.67.

Table 24. Table 2, reproduced from Nordhaus 2011.

| Base | 2015 | 2025 | 2035 | Low discount rate | 2015 | 2025 | 2035 |
|---|---|---|---|---|---|---|---|
| US | 3.60 | 4.38 | 5.28 | US | 10.93 | 13.63 | 16.47 |
| EU | 4.11 | 5.20 | 6.29 | EU | 7.73 | 9.91 | 12.00 |
| Japan | 0.78 | 0.95 | 1.11 | Japan | 2.07 | 2.58 | 3.07 |
| Russia | 0.51 | 0.79 | 0.95 | Russia | 1.25 | 1.85 | 2.24 |
| Eurasia | 0.48 | 0.87 | 1.24 | Eurasia | 1.22 | 2.00 | 2.72 |
| China | 10.40 | 23.92 | 31.70 | China | 28.94 | 57.03 | 74.05 |
| India | 7.98 | 16.91 | 26.03 | India | 20.11 | 37.17 | 53.13 |
| Middle East | 3.36 | 5.04 | 6.48 | Middle East | 8.98 | 12.98 | 16.32 |
| Africa | 7.83 | 13.87 | 24.75 | Africa | 29.62 | 47.17 | 72.84 |
| Latin America | 2.60 | 3.97 | 5.41 | Latin America | 6.87 | 10.00 | 13.11 |
| OHI | 1.37 | 1.77 | 2.06 | OHI | 4.17 | 5.43 | 6.44 |
| Other developing | 6.29 | 11.62 | 19.97 | Other developing | 26.45 | 43.87 | 67.59 |
| World | 41.49 | 62.50 | 83.56 | World | 134.38 | 198.59 | 261.89 |

**Table 3. Social cost of carbon by region, 2015-2035, base and low discount runs**

The social cost of carbon is measured in 2005 international US dollars. Countries' GDP are calculated using purchasing power parity exchange rates. To calculate the SCC per unit of $CO_2$, the figures should be divided by 3.67.

Table 25. Table 3, reproduced from Nordhaus 2011.