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February 2009

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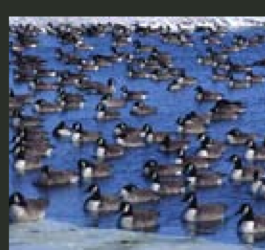

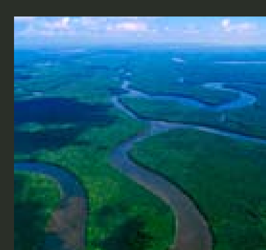

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This work was funded under Cooperative Agreement Number SRS 06-CA-11330139-222 between the USDA Forest Service Southern Research Station and Duke University.

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ABSTRACT

Under appropriate conditions, restoring wetlands on crop fields can result in a net increase of ecosystem services and therefore a net benefit to society. This study assesses the value of actions to restore wetlands via the Wetland Reserve Program (WRP) in the Mississippi Alluvial Valley (MAV) of the U.S. by quantifying and monetizing ecosystem services. Focusing on hardwood bottomland forest, a dominant wetland type of the MAV, *in situ* measurements of multiple ecosystem services are made on a land use continuum of agricultural land, wetlands restored via WRP, and mature bottomland forest. A subset of these services, namely greenhouse gas (GHG) mitigation, nutrient mitigation, and waterfowl recreation, are selected to be monetized with benefit transfer methods. Above- and belowground carbon estimates and changes in methane (CH_4) and nitrous oxide (N_2O) emissions are utilized to project GHG flows on the land. Denitrification potential and forgone agriculture-related losses are summed to estimate the amount of nitrogen prevented from entering water bodies. Increased Duck Energy Days (DEDs) on the landscape represent the WRP-induced expansion of waterfowl habitat. We adjust and transform these measures into per-hectare, valuation-ready units and then monetize them with prices from emerging markets (GHG) and environmental economic literature (GHG, nutrient, recreation).

Valuing all services produced by wetland restoration would yield the total ecosystem value of the change; however, due to data and model limitations we generate a partial estimate by monetizing three ecosystem services. Social welfare value is found to be between $1,446 and $1,497 per hectare per year, with GHG mitigation valued in the range of $162 to $213, nitrogen mitigation at $1,268, and waterfowl recreation at $16 per hectare. Limited to existing markets, the estimate for annual market value is merely $74 per hectare, but when fully accounting for potential markets, this estimate rises to $1,068 per hectare. The estimated social value surpasses the one-time public expenditure or social cost of wetlands restoration ($2,526 per hectare) in the MAV in only two years, indicating that the ecosystem service value return on public investment appears to be very attractive in the case of the WRP. Moreover, the finding that annual potential market value is substantially greater than landowner opportunity costs ($401–$411 per hectare) indicates that payments to private landowners to restore wetlands could be profitable for individual landowners in addition to being value-enhancing to society. This should help to motivate the development of ecosystem markets to more fully integrate societal values into land use decisions.

ACKNOWLEDGEMENTS

We would like to thank the U.S. Forest Service Southern Research Station for its support of this project, in particular the guidance and oversight of Dr. Patricia Flebbe. We appreciate the contributions made by David Shoch of TerraCarbon, Marc Ribaudo of USDA ERS, Ken Richkus of USFWS, Tom Moorman and Dale James of Ducks Unlimited, John Tirpak of the Lower Mississippi Valley Joint Venture, Lee Moore and Bruce McKenney of the Nature Conservancy, and participants in a workshop organized by the National Wetlands Research Center of the U.S. Geological Survey in Lafayette, Louisiana, in September 2007. We also are indebted to Marc Ribaudo for his insightful feedback on an earlier version of this report. Any remaining shortcomings in the study design are our own.

INTRODUCTION

In recent decades, U.S. agricultural policy has implemented programs that offer financial incentives to private landowners to spur restoration of natural habitat and its attendant ecosystem services. A younger sibling of the Conservation Reserve Program (CRP), the Wetlands Reserve Program (WRP) focuses specifically on the restoration, protection, and enhancement of wetlands on marginal farmland. Originally authorized in 1985, the acreage cap for WRP was expanded to 2.275 million acres in the 2002 Farm Security and Rural Investment Bill (USDA-NRCS 2007).

Ecosystem services, a collective term for the goods and services produced by ecosystems that benefit humankind, have traditionally been undervalued as they often fall outside of conventional markets and pricing (NRC 2005). Without market prices, the incentive to provide them privately has been low relative to other competing land uses, such as crops, timber, or mining. Furnishing evidence for this idea, the Millennium Ecosystem Assessment reported in 2005 that about 60% of global ecosystem services are being degraded or used unsustainably (MEA 2005). Increasingly, society has recognized the essential link between healthy ecosystems and human welfare and seeks ways to increase the provision of ecosystem services. Programs such as the WRP aim to stimulate provision of ecosystem services on private lands through strategic public payments to landowners and increased collaboration between landowners and government agencies. Also, substantial effort has gone toward the formation of nascent markets to allow the trading of new environmental commodities such as carbon offset credits (to mitigate greenhouse gases causing climate change) or water quality credits for land use actions that mitigate the introduction of nutrients and sediment to waterways. Economic valuation attempts to estimate the monetary values of these nonmarket ecosystem services so that they may be more fully accounted for in natural resource management decisions, both public and private.

An important dichotomy in economic values is that between social welfare value and market value. The first represents the economic value to society of the flow of ecosystem services and is the type of value which would be used in social benefit-cost analyses of public policies or programs. These social welfare values may pertain to varying geographical scales, as recreation is local, water quality is regional, and climate protection is global. Market value embodies what value landowners can capture through the market system and can be used to inform the design of landowner incentive programs for ecosystem protection or for the development of markets for ecosystem services. Market values encompass the goods, services, and assets traded in markets, ranging from traditional agriculture or land leasing to emerging commodities such as greenhouse gas (GHG) offsets.

This study focuses on the restoration of wetland ecosystem services in the Mississippi Alluvial Valley (MAV). The MAV covers the floodplain area below the confluence of the Mississippi and Ohio Rivers, principally located in the states of Arkansas, Mississippi, and Louisiana. Once containing nearly 10 million hectares (Mha) of bottomland hardwood forest, the MAV had only 2.8 Mha remaining by the 1980s following many decades of hydrological alteration and agricultural expansion (King et al. 2006). The major land use of the region is now agriculture, dominated by cultivation of corn, cotton, rice, and soybeans (USDA-NASS). This landscape transformation has had profound ecological consequences, such as wildlife habitat loss and fragmentation, loss of flood storage, and water quality degradation due to nonpoint source runoff.

The objective of the WRP is to restore and protect the functions and values of wetlands in agricultural landscapes with an emphasis on habitat for migratory birds and wetland-dependent wildlife, protection and improvement of water quality, flood attenuation, groundwater recharge, protection of native flora and fauna, and educational and scientific scholarship (USDA-NRCS 2004). The CRP has similar goals and objectives including improving the quality of water, controlling soil erosion, and enhancing wildlife habitat. The effectiveness of these conservation programs in achieving their goals and objectives, and thereby restoring ecosystem services, is not known for wetlands in the MAV. The USDA Conservation Effects Assessment Project (CEAP) began in 2003 as a multi-agency effort to quantify the environmental benefits of conservation practices used by private landowners participating in selected U.S. Department of Agriculture (USDA) conservation programs (Duriancik et al. 2008). As part of this program, the USDA CEAP-Wetlands component in the MAV has funded research on both natural forested wetlands and forested wetlands restored through the WRP and CRP. This research effort provides site-specific data on the ecosystem services supplied by these wetlands as well as by existing cropland. This data is used in valuation approach reported here.

This study aims to assess the value to society of actions to restore wetlands in the MAV. This objective is accomplished principally by comparing the economic values of ecosystem services produced on two land use types, agricultural land and restored wetlands. Constructing values from the bottom up, this study exploits a unique link between field data and economic valuation. Although the flows of ecosystem services are myriad, we confine ourselves to the three most well defined goods for the region’s wetlands: GHG regulation, nutrient retention, and waterfowl recreation. The findings of this analysis can provide valuable input into public and private decision making regarding natural resource management, including an assessment of the impact the WRP. Methodologies and values developed here will be available for use by other regional wetland assessments as well as more broadly for ecosystem service studies undertaken elsewhere.

RELATED LITERATURE

Advances in ecosystem sciences in recent years have increased our understanding of the critical importance that healthy ecosystems play in environmental sustainability. Because of human impact on ecosystems, efforts to maintain and restore ecosystems require an improved understanding of how humans benefit from ecosystems as well as how human behavior can be influenced through conservation payments and other policy tools (Heal 1991; Kramer 2008). A growing body of research has examined ecosystem services and their valuation, and government agencies are searching for ways to incentivize the provision of ecosystem services (U.S. EPA 2002; Ricketts et al. 2004; Barbier 2007).

Economists have been measuring ecosystem service values for years, for example, as part of legal proceedings to assess and assign natural resource damages from oil spills and other environmental accidents (Carson et al. 1994; NRC 2005). Enthusiasm for ecosystem services, however, expanded to the broader scientific and policy community due in part to two widely influential works published in the mid-90s by Daily (1997) and Costanza et al. (1997). Costanza's article sought to estimate the economic value of earth's ecosystems in their entirety. Most economists since then have followed the counsel of Toman (1998) to focus on changes in specific ecosystem service flows, as does this paper. In that vein, Loomis et al. (2000) measure the total economic value of the restoration of five ecosystem services for an impaired section of the South Platte River. Using contingent valuation, the authors find that households interviewed would be willing to pay $252 annually for this restoration and that scaling those values to all living along the river produces an aggregate benefit estimate that exceeds the water leasing costs and CRP easement costs needed to realize the restoration. Despite describing the environmental services in the survey, the WTP question treats them as a composite, making it impossible to decompose values for individual services. In contrast, Chan et al. (2006) implement a conservation-planning framework to examine trade-offs between biodiversity and six other ecosystem services, but do not attempt to value the services economically. Their approach reveals spatial correlations between biodiversity and the production of ecosystem services and provides information on the relative impacts of different conservation targets on those services.

Two recent articles have conducted statistical meta-analyses of wetland valuation studies, using wetland value per unit area as the dependent variable. Woodward and Wui (2001) draw data from 39 studies, predominantly of temperate wetlands, while Brander et al. (2006) use 80 studies from 25 countries representing all the continents. Updating to 2008 U.S. dollars, the former found a mean annual value per hectare of $567 among its constituent studies, whereas the latter computed a mean of over $4,000/ha/yr but a median of $215. Significant decreasing returns to scale are noted as wetland area grows in both analyses, though Woodward and Wui (2001) assert that area has a minimal impact on value per acre because this effect rapidly approaches zero with increasing wetland size. Regarding the values of different wetland services, only bird watching (Woodward and Wui) has significantly higher value than average, while bird hunting and amenity services (Woodward and Wui) and hunting, material, and fuelwood services (Brander et al.) are found to be significantly lower than average. In each meta-analysis, the service nutrient retention is classified under water quality and GHG mitigation is not included at all. Both studies conclude that benefit transfer still faces major challenges and that the need for more high-quality primary valuation studies continues to be great.

A few studies have examined the benefits associated by the Conservation Reserve Program (CRP). Feather and Hellerstein (1997) evaluate the national benefits of reduced soil erosion for recreation by estimating the benefits in four study areas and then extrapolating them to the nation as a whole with a calibration function that accounts for area-specific factors. The authors report that 11%, or about $40 million, of the nationwide benefits are attributable to the CRP. Surveying both nationally and in Iowa, Ahearn et al. (2006) find that a conservative non-use value of the Central Plains grassland birds that increase in numbers due to the CRP to be about $33 million per year.

Anderson and Parkhurst (2004) consider farmers’ decisions to continue commodity crop production or to enroll in the Wetland Reserve Program (WRP) in the Mississippi delta region. In their study, land was more likely to be entered into WRP if its crop base was soybeans/soybeans or cotton/soybeans and if it had considerable recreational value. In a similar analysis, Ibendahl (2008) simulates the farmers’ decisions for three counties in Mississippi using crop budgets for 2008 which reflect the historically high crop prices. He concludes that the 30-year stream of crop returns and government payments for cotton or soybean production exceeds the expected per-acre WRP payment.

ECOSYSTEM SERVICE CONCEPTUAL MODEL

We are interested in estimating the value of ecosystem services associated with a change in the use of a given unit of land. Land is an asset that generates a flow of different services.

Some of the flow is in biophysical outputs that are directly sold in the agricultural market and perhaps the timber market. Other flows work though a series of ecological and spatial processes before they become part of a service that can be valued. For instance, nutrient retention is not a valued service per se; it becomes a valued service only after working through the hydrological system to create a change in water quality. Likewise, there can be complex relationships between the existence of a unit of a particular habitat in the area of interest and its relationship to what people value either locally or at a distance.

To describe the valuation process, we start with basic hedonic model (Rosen 1974; Palmquist 1989) of value, *V*:

$$V = V(a) \qquad [1]$$

where *a* = a vector of site attributes (e.g., size, soil quality, elevation, infrastructure, population, proximity to markets).

The ecosystem service flows are reflected in a vector, *S*, that is a function of the underlying attributes

$$S = S(a) \qquad [2]$$

The service vector *S* has three subvectors:

S_M(a): goods and services that can be sold in markets, (e.g., agricultural and forest commodities, housing, marketed ecosystem services such as hunting)

S_C(a): in situ goods and services consumed by the owner of the land (e.g., residential space, nonmarketed products, amenity values)

S_P(a): services that generate public goods that do not (yet) have markets (e.g., nutrient retention, biodiversity)

It should be noted that some of these services can be produced simultaneously on the same plot of ground (e.g., commodities and certain ecosystem services), while others require explicit choices and cannot be co-produced.

Hence, the flow value of land is expressed as the sum of the value of market and nonmarket services generated:

$$V(S) = p^*S_M + v^*S_C + w^*S_P \qquad [3]$$

where p is a vector of market prices matched with the market good/service vector, v is a vector of implicit prices reflecting the values of each self-consumed good/service, and w is a vector of implicit prices reflecting the marginal value to society of the public good/service vector generated onsite.

The market value of the land (rental) should reflect the array of market services generated in highest and best use. In other words, the prices of market goods and services and self-consumed goods will determine how the landowner chooses the level of market/consumed services that will be generated by the land (how much of marketed commodity, how much residential space, etc.). Hedonic value, as a function of attributes, is a reduced-form version of that $V = V(a)$. In other words, the site attributes are deemed to dictate the choices that determine the "highest and best use."

Hedonic models usually try to capture the relationship between market data (property values, which are a capitalized expression of the value flow, V) and attributes (a) to give marginal values of each. But here, given that there are no markets for the ecosystem services except those that have a market price or are self-consumed (in vectors S_M and S_C), hedonic valuation cannot help us determine ecosystem service values generated by the land. Because the market value does not capture all value, the market does not allocate to highest and best use. If all ecosystem services were valued in the market, then in principle it could.

So we can examine comparative values across discrete uses and see how optimal land allocation might occur if the market valued it (or if there were government intervention with payments for ecosystem services).

We are specifically interested in testing the hypothesis that the change in total economic land value increases as one changes from agriculture to wetlands:

$$H_O: V_W(a) > V_A(a) \qquad [4]$$

where $V_W(a)$ is the total value of land, inclusive of all ecosystem services whether marketed or not, when it is in wetlands and $VA(a)$ is the total value of land in agriculture.

As an economic principle, we believe that if land is in agriculture, then the sum of all marketed and self-consumed services in agriculture must be higher than the sum of all marketed and self-consumed services in wetlands, or any other use. The real issue, then, is whether the difference in public goods value exceeds the difference in market value.

Before proceeding, we acknowledge there are criticisms leveled at this "total economic value" approach to ecosystem services stemming from the fact that the estimated value is the sum of all measured services times their shadow price (see Howarth and Farber 2002 for a review of the arguments). The critical issue is whether it is reasonable to assume the shadow price remains fixed when the ecosystem service quantity changes. In standard economics terms, it is a matter of using a partial equilibrium approach for a general equilibrium problem. This is clearly problematic when the stock value of entire ecosystems is being valued, as presumably large changes in these services are at issue and prices (marginal values) would have to change. We do not believe this is a significant problem for this study. First, we are looking at changes in ecosystem services brought about by marginal changes in land use, not at the existence of entire ecosystems. The WRP, while an important public program, does not change the landscape at a scale large enough to fundamentally alter demand for the various services, and therefore has not likely changed the shadow prices either, or if they have changed, the change is small. Therefore, in our view, a more general equilibrium approach is not needed. However, one should be careful in interpreting the implications of these results for changes of a larger magnitude.

APPLICATION

Study Area

The Mississippi Alluvial Valley (MAV) is the nation's largest floodplain, extending from below the confluence of the Mississippi and Ohio Rivers to southern Louisiana (Figure 1). About three-quarters of the original bottomland hardwood forests have been converted, principally to row crop agriculture, while the remaining quarter is fragmented into over 38,000 discrete patches larger than 2 ha in size (Twedt and Loesch 1999). The study area encompasses all of the counties that intersect with the MAV, save for those in Louisiana bordering the Gulf of Mexico.

Figure 1. Extent of the Mississippi Alluvial Valley (MAV) and the locations of the 16 WRP sites sampled by USGS scientists.

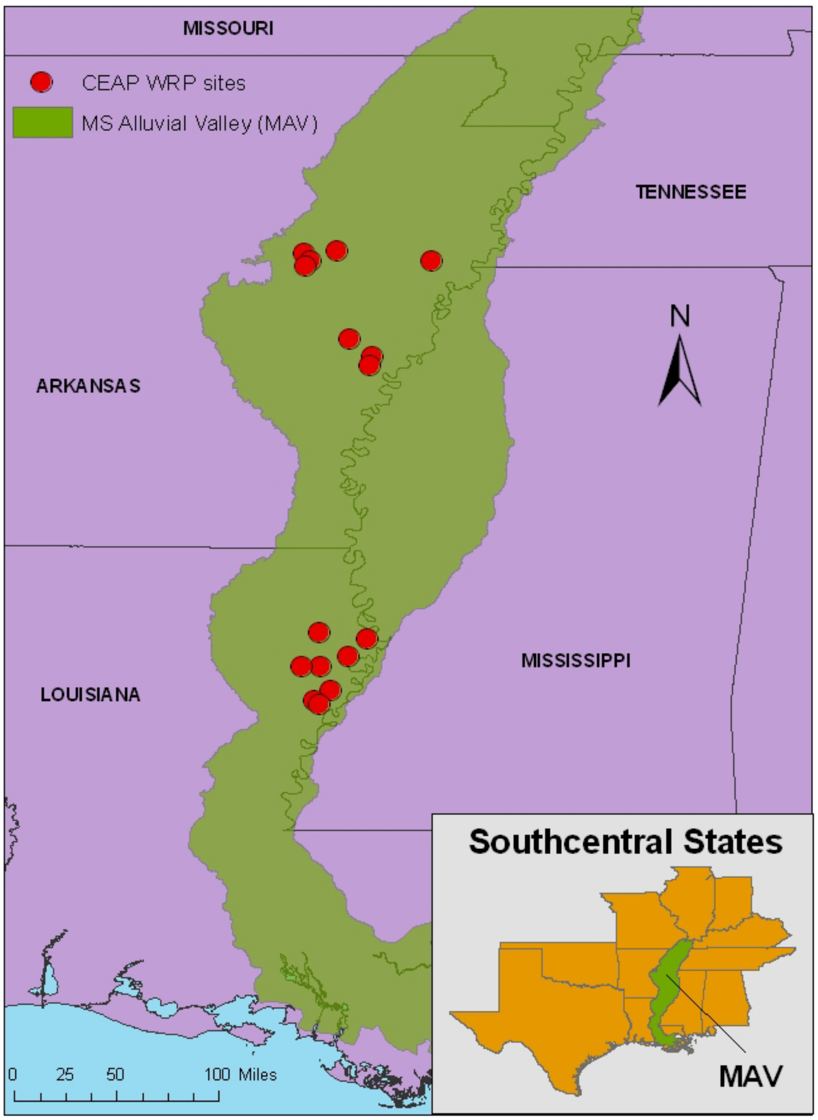
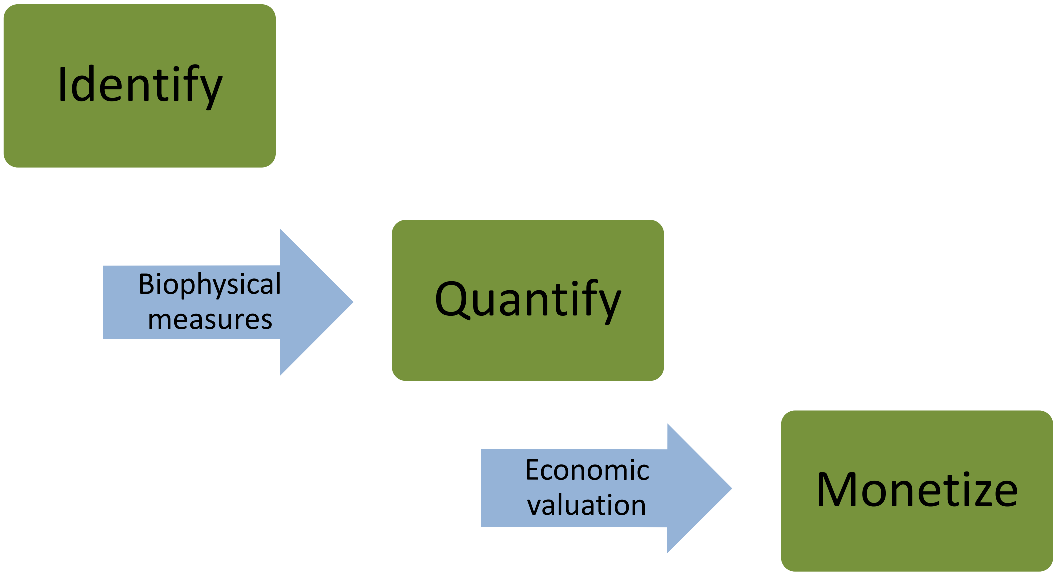


Figure 2. Flow chart of ecosystem service valuation process.



Benefit Valuation Process

There are three essential steps in the ecosystem service valuation sequence: (1) identify the service, (2) quantify the service flows, and (3) monetize those flows (Figure 2). Disciplines that assess biophysical processes, such as ecology, biogeochemistry, and hydrology, play the central role in moving from identification to quantification. Economics then provides the link from service quantification to monetization. Critical to bridging the biophysical and human aspects of ecosystem services is to transform the service flow data into valuation-ready measures. This transformation may involve integrating field observations with existing process models and modeling the service through time. We standardize the service measures into per-hectare values to facilitate comparisons with economic returns from other land uses and the aggregation of benefits to broader scales. Using benefit transfer methods (Wilson and Hoehn 2006), we multiply biophysical values for services of interest by shadow prices for the services (see conceptual model discussion). These prices are obtained either through market price observations or from estimates of marginal willingness to pay for these services in the environmental economics literature. We focus on the monetization of three services: GHG mitigation, nitrogen mitigation, and waterfowl recreation, which prior information suggests are the dominant service flows for the MAV region in terms of economic value.

Although new ecosystem markets are emerging, ecosystem services can generally be considered public, nonmarket goods. When valuing a nonmarket good, total economic value (TEV) is the sum of use values, which are directly or indirectly derived from the use of an ecosystem, and nonuse values, which are related to the ecosystem's existence (Krutilla 1967; Young 2005). Thus, the TEV is equivalent to the monetization of the flow of the services from an ecosystem. In the conduct of primary research, nonmarket valuation approaches tend to be divided into two main categories: (1) stated preference and (2) revealed preference (Freeman 2003). Stated preference methods use data of intended behavior derived from survey questions directly asking respondents how they would value differing levels of an environmental good. Contingent valuation and conjoint analysis are two examples of stated preference methods. Revealed preference methods utilize observed market prices, travel costs, and purchase decisions that are correlated with changes in an environmental attribute as indicators of value for that attribute. Examples include observed market prices for some services (e.g., GHG reductions, hunting leases), travel cost method for recreation values, hedonic property value studies, and estimation of avoided expenditures to achieve a certain level of an environmental attribute (e.g., water quality).

Acknowledging that time and resources are scarce, the benefit transfer method builds on the previous methods by applying results from primary research to new contexts of interest (Rosenberger and Loomis 2003). For example, the benefits estimated for a water quality improvement in one region may be adapted to estimate the benefits of an improvement in another region. A proper benefit transfer requires that the original study site be comparable to the targeted policy site with respect to the ecosystem service definition, the market (i.e., human population) context, and the welfare measure employed (Loomis and Rosenberger 2006).

In each application in this analysis, agricultural land use is treated as the baseline, since it represents the dominant land use in the MAV, and thus the business-as-usual scenario prior to restoration. Seeking to value the action of restoring forested wetlands on cropland, we capture this economic value by calculating the difference in the values of ecosystem services provided by the two respective land use types.

Biophysical Measurement of Ecosystem Service Flows

Scientists at the USGS National Wetlands Research Center carried out the sampling design and the data collection for this study as part of the CEAP-Wetlands component (Faulkner et al. 2008). Initiated in 2003, CEAP is a multi-agency effort to evaluate the effectiveness of conservation practices used by private landowners participating in selected USDA conservation programs (USDA-NRCSa). A major element of CEAP is the National Assessment, whose objectives are to collect national estimates of benefits resulting from conservation practices and programs for croplands, wetlands, wildlife, and grazing lands and to weigh the potential of existing and future conservation programs to meet the nation’s environmental goals. The wetlands component of the National Assessment measures the effects of conservation practices on ecosystem services provided by wetlands in agricultural landscapes and is being conducted in eleven regions throughout the coterminous U.S. These regional assessments will focus on one or more wetland hydrogeomorphic classes common to agricultural land in that region.

For the CEAP-Wetlands study in the MAV, a stratified random sampling design was used in the Lower White-Cache and Tensas river basins where eight replicate sites were selected for each of three treatments: restored to forested wetlands under the WRP, active cropland, and natural forested wetland sites. These sites are representative of the variability on the landscape and add up to 48 sites in total, 16 each of cropland, WRP, and natural forest. Site-level field data was collected between March and October 2006 for four ecosystem services, while soil samples for the denitrification measurements were taken in

2007. Three involve biogeochemical processes, namely, carbon sequestration, nutrient retention, and sediment retention, and the other two involve biological conservation, i.e., amphibian species richness and neotropical migrant bird species richness. Region-level data for migratory waterfowl habitat was calculated by estimating the extent of flooding based on Landsat Thematic Mapper (TM) classified image analysis for 2000–2005 and the estimated waterfowl foraging values of reforested areas (James et al., in review). Using the static chamber technique, methane and N_2O emissions were measured monthly from low- and high-elevation sites in both WRP and natural forested wetlands from 2005–2008 at 18 sites in the MAV different from the CEAP-WRP sites (Faulkner, unpublished data). Table 1 lists the relevant services with the metric measured and its spatial resolution.

Table 1. Ecosystem services measured by USGS National Wetlands Center and Ducks Unlimited.

Ecosystem Service	Definition/Metric	Spatial Resolution
Wildlife habitat – amphibians	Species richness (number/ha)	Site
Wildlife habitat – breeding birds	Species richness (number/ha)	Region
Wildlife habitat – waterfowl	Duck energy days/acre	Region
Nutrient retention	Denitrification potential (kg NO_3-N/ha/yr)	Site
Erosion reduction	Sediment (Mt/ha/yr)	Site
Carbon sequestration	Mg CO_2e/ha/yr	Site

ECOSYSTEM SERVICE VALUATION

Greenhouse Gas (GHG) Mitigation

Converting land from croplands to forested wetlands can affect the GHG balance in the atmosphere in several ways. First, carbon dioxide (CO_2), the most prevalent GHG, is removed from the atmosphere via photosynthesis and is sequestered in forest biomass and soils at levels typically well above the sequestration rate for crop systems. This creates a net carbon sink and reduces GHG concentrations, all else being equal. Second, crop production can be a significant source of non-CO_2 trace GHGs such as nitrous oxide (N_2O) and methane (CH_4), gases that are individually more potent than CO_2. Thus, discontinuation of agricultural practices reduces these emissions from the site. However, the anaerobic conditions of wetlands are ideal for the creation of methane and nitrous oxide and thus conversion can increase emissions accordingly. The net balance is determined by site conditions, as discussed below.

The process of converting GHG biophysical measures to monetary values is described below for carbon sequestration and non-CO_2 GHGs respectively.

Carbon sequestration

The biophysical data collected by the CEAP research team for this service are point estimates of aboveground and soil carbon in metric tons of carbon per hectare in the first few years after restoration. Because carbon accumulation in ecosystems is a dynamic process, these point estimate snapshots need to be transformed into GHG flux over time in order to be properly monetized. Carbon accumulation growth is tracked in three carbon pools—soil, live biomass, and other non-soil—and is projected for the future employing two different process models.

Soil carbon

For soil carbon sequestration, we average the soil carbon point estimates to create mean carbon values for all sites in each land use class (cropland, WRP land, and mature forest). Site soil carbon data are provided for the upper 15 cm of soil, where soil carbon is highest before decreasing dramatically with depth. These data are a fair proxy for one meter of soil depth, the standard used in soil carbon estimation. Next, we seed the WRP mean values, 20.83 Mg[1] C/ha/yr for Arkansas and 24.07 for Louisiana, into stand-level tables developed by the U.S. Forest Service as part of the federal 1605(b) GHG registry process. These tables are derived from the FORCARB2 forest carbon projection model (Smith et al. 2006). These tables contain data on carbon accumulation growth paths for afforested and reforested stands in 5-year increments by carbon pool, forest type, and U.S. region. To use the FORCARB2 soil model, WRP land in the MAV is proxied by afforested oak-gum-cypress forest in the south-central U.S. The growth paths are

[1] The abbreviation *Mg* stands for *megagram*; 1 Mg is equivalent to 1 metric ton (tonne) or 10^6 grams. This paper uses Mg except in the context of the carbon credit trading market, in which the standard abbreviation *tCO_2e* is used to refer to "metric tons of CO_2 equivalent."

traced out in 5-year time steps for 90 years from the initial year of restoration (see Table 2). Soil organic carbon at WRP sites is assumed to follow the same growth path as reported in the FORCARB2 lookup tables, though the beginning value is that provided by the CEAP field data.

Table 2. Growth and net carbon flux over 90 years in soil organic carbon for agricultural and WRP sites in Arkansas and Louisiana.

	FORCARB2 table	CEAP Data – AR		CEAP Data – LA		AR		LA	
		Ag	WRP	Ag	WRP	Ag	WRP	Ag	WRP
Age	Soil Organic Carbon					Carbon Flux			
yrs	Mg C/ha					Mg C/ha			
0	29.00		–		–				
5	29.10	20.80	20.83	21.84	24.07	0.00	0.00	0.00	0.00
10	29.40	20.51	21.05	21.54	24.31	−0.29	0.21	−0.29	0.25
15	29.80	20.23	21.33	21.24	24.64	−0.28	0.29	−0.28	0.33
20	30.40	19.95	21.76	20.95	25.14	−0.28	0.43	−0.28	0.50
25	31.10	19.68	22.26	20.66	25.72	−0.27	0.50	−0.27	0.58
30	31.90	19.41	22.84	20.38	26.38	−0.27	0.57	−0.27	0.66
35	32.70	19.14	23.41	20.10	27.04	−0.27	0.57	−0.27	0.66
40	33.50	18.88	23.98	19.82	27.70	−0.26	0.57	−0.26	0.66
45	34.30	18.62	24.55	19.55	28.37	−0.26	0.57	−0.26	0.66
50	35.10	18.36	25.13	19.28	29.03	−0.26	0.57	−0.26	0.66
55	35.80	18.11	25.63	19.01	29.61	−0.25	0.50	−0.25	0.58
60	36.40	17.86	26.06	18.75	30.10	−0.25	0.43	−0.25	0.50
65	36.90	17.61	26.41	18.49	30.52	−0.25	0.36	−0.25	0.41
70	37.30	17.37	26.70	18.24	30.85	−0.24	0.29	−0.24	0.33
75	37.60	17.13	26.92	17.99	31.10	−0.24	0.21	−0.24	0.25
80	37.90	16.90	27.13	17.74	31.34	−0.24	0.21	−0.24	0.25
85	38.10	16.66	27.27	17.49	31.51	−0.23	0.14	−0.23	0.17
90	38.30	16.43	27.42	17.25	31.67	−0.23	0.14	−0.23	0.17

At the agricultural sites, the initial soil carbon values come directly from the agricultural sites paired with the WRP sites in Arkansas and Louisiana. Conventional tillage is the assumed agricultural practice. In contrast to the WRP sites, agricultural soil carbon levels tend to gradually decrease over time as they are oxidized and released into the atmosphere as a result of crop production (Potter el al. 2006a). A 2006 NRCS study simulates the change in soil carbon content for agricultural lands over a 30-year time period with the Environmental Policy Integrated Climate (EPIC) model (Williams et al. 1989; Potter et al. 2006b). The analysis provides soil organic carbon estimates, as well as those for soil and nutrient losses, by region and by crop type.

Live biomass carbon

The non-soil carbon data from CEAP represents aboveground and belowground (i.e., coarse roots) live carbon biomass plus standing dead, understory, and forest floor carbon. Across the WRP forested wetland sites that had been planted between 4 and 12 years prior to sampling, non-soil carbon measurements average 2.70 Mg/ha in Arkansas (1.69–6.33 Mg/ha range) and 3.06 Mg/ha in Louisiana (1.79–5.71 Mg/ha range).

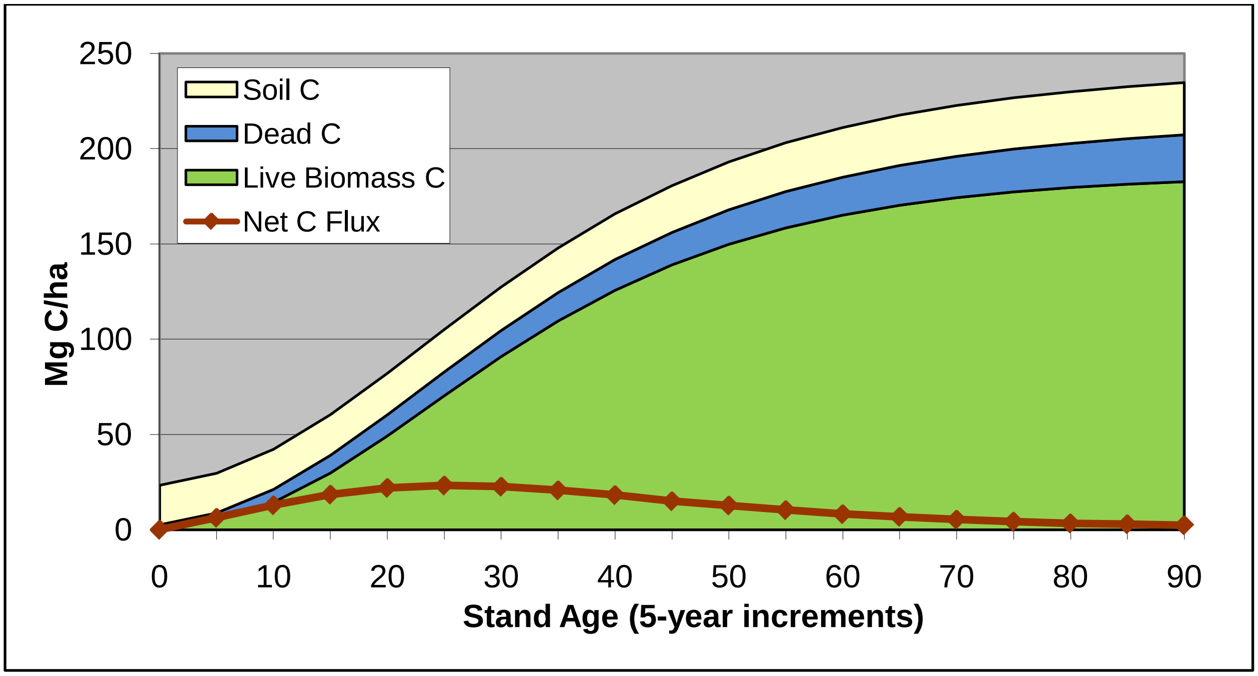
The majority of carbon sequestration potential resides in the growth of live carbon biomass (e.g., trees) through time, increasing from 72% at year 10 to over 86% in year 90 according to the USFS FORCARB2 tables (Smith et al. 2006). We estimate the carbon accumulation flows of this pool using the growth function from Shoch et al. (2008) who examine the carbon sequestration potential of bottomland hardwood afforestation in the MAV. The authors produce a chronosequence of even-aged plantations and naturally regenerated stands and statistically estimate a growth path that is markedly greater for years 20 to 90 than that derived from the USFS FORCARB2 tables for afforested oak-gum-cypress stands (Smith et al. 2006), which are commonly used for regional analysis.

This substantial difference between Shoch et al. and FORCARB2 is neither surprising nor a criticism of the FORCARB2, which is clearly defined as a model with large regional resolution. The estimated growth curve from Shoch et al. is specific to the MAV and is thus more appropriate for our study than the FORCARB2 tables whose estimates are for the south-central region in general. Dominated by bottomland red oaks, stem plantings in the WRP sites are very similar in species composition as the plantations surveyed by Shoch et al. (2008), further validating the use of their growth function. The CEAP field data for non-soil carbon falls approximately within the 95% confidence interval of and well within the prediction interval of the total live tree biomass carbon growth curve generated by Shoch et al. (2008). Therefore, it is appropriate to project future live tree carbon accumulation for the WRP sites with the Shoch et al. (2008) growth function.

Other carbon

MAV-specific estimates for carbon found in standing dead, understory, and forest floor (i.e., not found in live trees) are currently unavailable, so we utilize the USFS FORCARB2 tables as the best available source. Growth in carbon in those pools is projected in the same way as described above for the WRP soil organic carbon. In Figure 3, the carbon accumulation curve is depicted, with each major carbon pool represented by a different colored area.

Figure 3. Carbon growth and net carbon flux curves for afforested bottomland hardwood on WRP sites in Mississippi Alluvial Valley.



Carbon flux (Mg C/ha/time period) is the net change of carbon on the site from one period to the next so that positive carbon flux represents new carbon stored in addition to the existing carbon stock. This is the service flow of interest as it directly relates to the removal of CO_2 from the atmosphere, which provides the climate stabilization benefit. Flux often varies through time following the growth rate of the vegetation and soil carbon storage. The projected carbon flux for the WRP sites is represented by the red line in Figure 2. Agricultural sites (not shown here) have a slightly negative carbon flux, since soil carbon declines gradually from soil oxidation associated with crop production (Potter et al. 2006a) and the biomass grown in crops each year is also removed from the land on an annual basis. Once the carbon fluxes for total site carbon have been calculated for the agriculture and WRP sites, we then convert them into units of carbon dioxide equivalents (CO_2e) by simply multiplying by 3.67. CO_2e is the currency in which carbon service flows are monetized.

Non-CO_2 GHG emissions

The last step in quantifying the GHG sequestration potential is to account for the effect of emissions of trace GHGs, methane (CH_4), and nitrous oxide (N_2O). They have global warming potentials (GWP) much greater than CO_2 itself: 23 for CH_4 and 296 for N_2O (IPCC 2007). Both crop and wetland sites are net sources of CH_4 and N_2O emissions, though of different magnitudes. Accordingly, site N_2O and CH_4

fluxes are converted to their CO_2 equivalents using the GWP above, and are then subtracted from the CO_2 flux to determine the net GHG flux ($MgCO_2e/ha/yr$).[2]

For the agricultural sites in the region, CH_4 is emitted through rice production and residue burning and N_2O is emitted through the use of nitrogenous fertilizers and nitrogen fixation by soybeans. To find these GHG fluxes, we first determine the crop mixes for a representative agricultural hectare in the MAV for each state using data compiled by USDA National Agricultural Statistics Service (USDA-NASS). Then, we multiply the crop mixes by the corresponding state average estimates for agricultural CH_4 and N_2O emissions from the FASOMGHG model (Adams et al. 2005). Finally, weighted averages for the three MAV states are produced: −5.51 $MgCO_2e$ /ha/5 years for CH_4 and −3.14 $MgCO_2e$ /ha/5 years for N_2O.

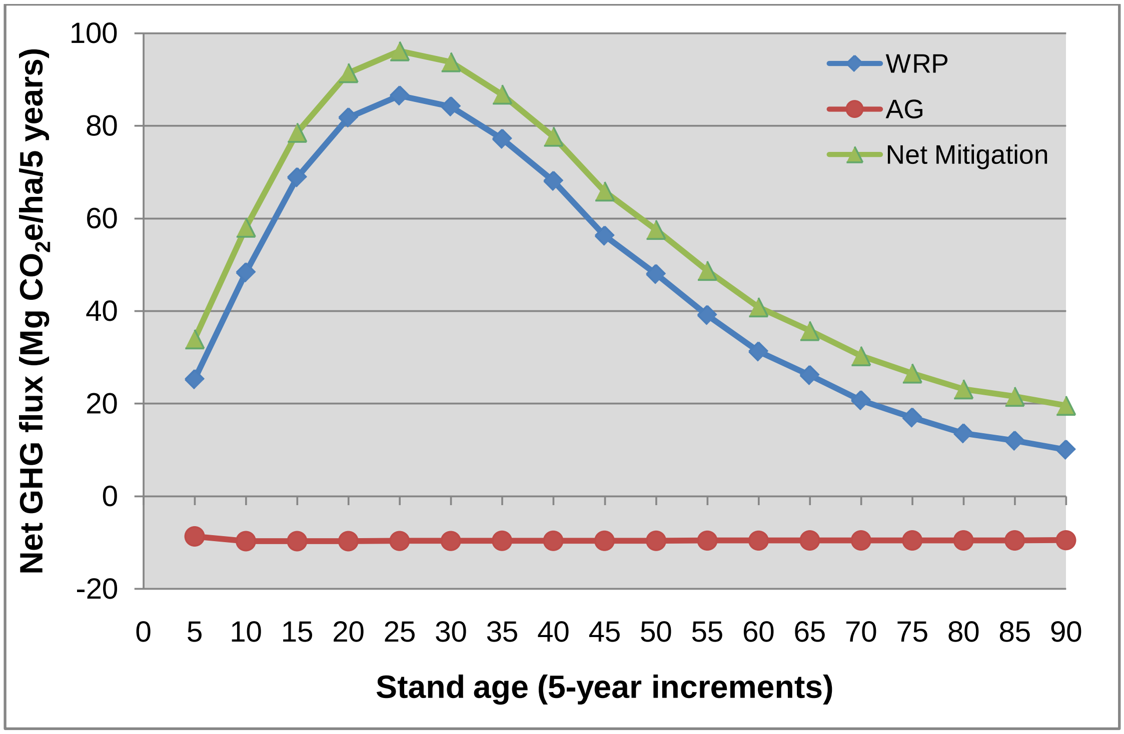
For both WRP and natural wetland sites, the levels of CH_4 and N_2O emissions vary by landscape position, i.e., whether the site is located in a low- or high-elevation position. Low-elevation sites flood more frequently and for longer duration than high-elevation sites and thus will experience longer periods with anoxic conditions in the soil. This anoxia is a prerequisite for the processes of methanogenesis and denitrification to produce gaseous methane and convert nitrate into gaseous dinitrogen (N_2) and nitrous oxide (N_2O) (Mitsch and Gosselink 2007). Since the goal of WRP is to remove frequently flooded, marginal croplands from commodity crop production, we estimate that approximately 80% of the WRP area is characterized by low elevation and the other 20% by high elevation. We multiply the CH_4 and N_2O emission rates for each landscape position by the corresponding proportion (0.8/0.2) and generate a weighted average of CH_4 and N_2O emissions for each 5-year increment between years 5 and 90 after the wetlands restoration. After converting to $MgCO_2$ equivalents, the mean CH_4 flux is −0.13 $MgCO_2e$/ha/5 years and the mean N_2O flux was −2.02 $MgCO_2e$/ha/5 years.

Total GHG flux change

Since a typical agricultural site candidate for restoration serves as the baseline, full GHG flux for restoring a hectare of wetland is the difference between the GHG fluxes for the average MAV agricultural and WRP sites. Figure 4 shows these three flux streams over the 90-year study period. Agricultural sites function as sources of GHG emissions and have a negative flux value for mitigation purposes (see footnote 2). In contrast, WRP sites serve as net sinks, have a positive mitigation flux value, and sequester up to 84 Mg of new CO_2 per hectare per 5-year period. Although non-CO_2 GHG gases are emitted in restored wetlands, their contribution is easily offset and exceeded by the carbon sequestration of the growing wetland forests. The net GHG mitigation value of restoring wetlands ranges between 19.6 and 96.2 Mg CO_2e/ha/5 years, with the peak coming at 25 years after planting the tree seedlings.

[2] We depart from some convention on the sign of the flux. We use the terrestrial ecosystem itself as the stock from which fluxes occur. Thus, a negative flux is an emission (e.g., release of CO_2 from oxidized soil carbon or the release of N_2O from denitrification), whereas carbon sequestration is a positive flux. We do this to highlight the notion that a positive number (increased sequestration or reduced emissions) is “mitigation” representing an environmental benefit that can receive a positive payment as discussed throughout.

Figure 4. Net greenhouse gas (GHG) mitigation from converting agricultural sites (AG) to WRP sites.



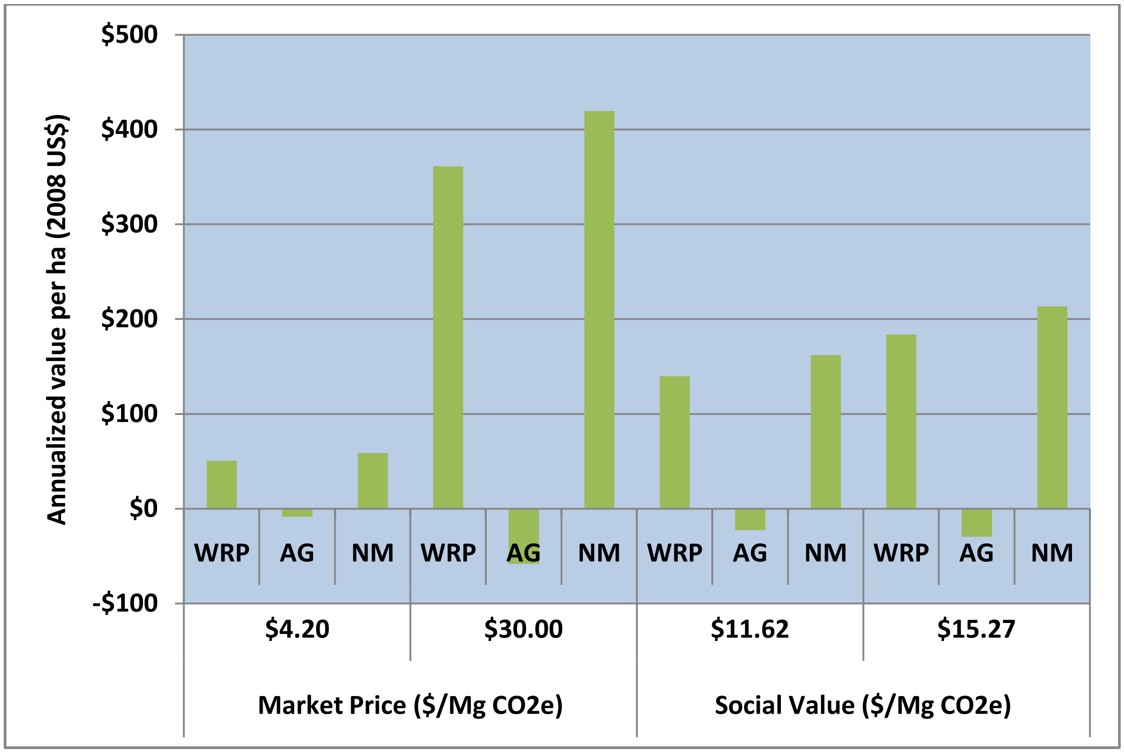
Monetizing GHG mitigation

The social welfare value of GHG mitigation captures the value of the damages avoided by mitigating the risks of climate change. This is typically estimated with the use of integrated assessment general equilibrium models to capture the *social cost of carbon*, or SCC. The IPCC Fourth Assessment Report (2007) reviews studies in the environmental economics literature that investigated the benefits of GHG mitigation and finds that mean estimates for SCC range from about \$12/MgCO_2 to \$15/MgCO_2. We use this as the shadow price for 1 Mg of GHG mitigated on our study sites.

Present value calculation

The stream of total GHG flux per hectare is multiplied by the market and social value prices and then discounted back to the present with a 4% real discount rate. The net present value of the GHG mitigation service is divided by the 90-year annuity factor to yield the annualized values per hectare that appear in Figure 5. Note that the discussion of how we determined the range of market prices used here is found further on in the Market Value section. The monetized net mitigation value is the difference between the WRP and agriculture sites. It ranges from \$59/ha/yr to \$419/ha/yr for the market prices of \$4.20 and \$30.00 respectively, while the social values are intermediate at \$162/ha/yr to \$213/ha/yr.

Figure 5. Annualized value per hectare in 2008 US$ for WRP, agricultural sites (AG), and net mitigation (NM) under market and social value prices for $MgCO_2e$.



Nitrogen Mitigation

Quantifying nitrogen service flows

Nitrogen is a major nutrient in agricultural runoff linked to water quality degradation in general (Carpenter et al. 1998) and, specifically, its increase in loading to the Mississippi River is considered a principal cause of the hypoxic "dead zone" in the Gulf of Mexico (Goolsby and Battaglin 2001). There are two principal ways in which wetlands restoration mitigates environmental damage from nitrogen releases: (1) forgone nitrogen (N) losses associated with runoff from crop cultivation and (2) removal of nitrate (NO_3) via denitrification.

When land is enrolled in a WRP easement, it is by definition taken out of agricultural production and thus the N losses driven by fertilizer application, fixation, and tilling cease. Because nitrate is the species of N most clearly correlated with the hypoxic zone size in the Gulf of Mexico, we focus on nitrate loading in our analysis (Mississippi River/Gulf of Mexico Water Nutrient Task Force 2007). We compute the nitrate prevented from entering the local waterways by applying average annual values for nitrate lost in surface water runoff, in lateral subsurface flow, and in leachate (N kg/ha/yr) from agricultural sites using output from the EPIC model (Potter et al. 2006b). These EPIC model estimates are available by U.S. region and by primary crop type within each region (Potter et al. 2006a). Knowing the counties in which the paired WRP and reference agricultural sites are located in the MAV but not their exact location due to privacy restrictions, we create representative crop sites for the MAV portion of each state with USDA data that details the crop mix for those counties (USDA-NASS). The nitrogen loss estimates for each crop type are combined with the crop type proportions to produce total nitrogen loss for a representative agricultural hectare in the MAV in that state. See Table 3 for an example calculation for Arkansas. Total nitrate ground- and surface-water losses for the MAV counties in Arkansas, Louisiana, and Mississippi are 41.3, 29.3, and 32.3 kg/ha/yr, respectively. Computed using the relative total hectares planted in crops in the MAV counties for each state, the weighted average of agriculture-related N loss for the MAV is 37.0 kg/ha/yr.

Table 3. Estimated nitrate loss by crop type from a representative agricultural hectare in the MAV in Arkansas.

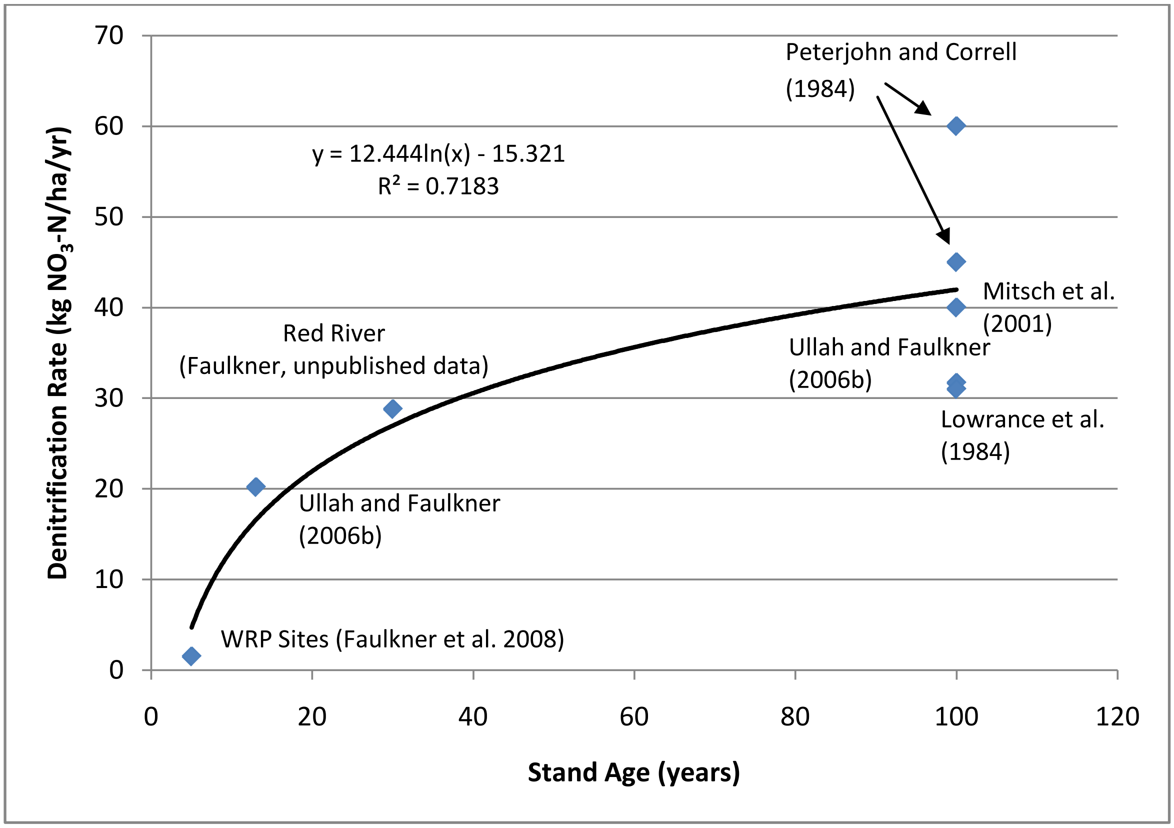
Crop Type	Estimated NO_3 Loss	Crop Type	Crop Contribution
	kg/ha	Proportion	kg/ha
Corn	24.9	0.031	0.8
Cotton	29.4	0.1	2.9
Rice	69.9	0.32	22.4
Sorghum	13.1	0.005	0.1
Soy	29.0	0.516	15.0
Winter Wheat	5.7	0.028	0.2
		Total	**41.3**

The second mitigation pathway is the removal of nitrate (NO_3) through the denitrification process, which is the primary N loss process in freshwater wetland ecosystems (Faulkner and Richardson 1989; Mitsch et al. 2001). The complex interactions of hydrology, soil type, nutrient loadings, and landscape position create the variability in specific ecosystem processes found in natural wetlands (even within a wetland type) and there is a wide range in reported nutrient retention rates due to differences in specific processes controlling those rates (Faulkner and Richardson 1989; Reddy et al. 1999; Novak et al. 2004; Lowrance et al. 2006). Reported denitrification rates in natural forested wetlands range from <1 to >800 kg N ha^{-1} y^{-1} (Mitsch et al. 2001, Lowrance et al. 2006). In addition, there is evidence that restored forested wetlands have different rates that change as the system ages and develops ecosystem characteristics more similar to forests than croplands (Hunter and Faulkner 2001; Ullah and Faulkner 2006a). This variability makes it difficult to predict N retention rates for WRP sites through time. We estimated denitrification potential (kg NO_3/ha/yr) with the denitrification enzyme assay (DEA) using field soil samples from both cropland and WRP CEAP sites. This denitrification potential approximates the rate at which nitrate is removed by the site. The DEA is a widely used approach (Groffman and Tiedje 1989; Clement et al. 2002; Ullah and Faulkner 2006a). We also reviewed published denitrification rates and found several studies that were similar to the WRP and natural sites evaluated here (Peterjohn and Correll 1984; Lowrance et al. 1984; Mitsch et al. 2001; Ullah and Faulkner 2006a, 2006b).

In order to capture the future denitrification potential of the restored wetlands, we modeled the relationship between the ages of forested wetland stands and the denitrification rates using the CEAP

WRP data; unpublished data from sites at Red River, Louisiana; and six point estimates from the literature. As can be seen in Figure 6, a log function fits the data well with a R^2 value of 0.7183. We use this curve to represent the age-dependent trajectory of denitrification through the 90-year study period at sites with a low landscape position. Since none of the published denitrification rates distinguish between low- and high-elevation sites in forested wetlands, we used experimental data which indicates that high-elevation sites display denitrification rates that are about 10% of those of low-elevation sites—low 28.8 kg N/ha/yr vs. high 2.88 kg N/ha/yr (Faulkner, unpublished data). Therefore, we assume that denitrification rates at high-elevation sites have the same trajectory as those at low-elevation sites, but with one-tenth of the value. Applying our assumption that 80% of the area of the WRP sites is low-elevation and 20% is high-elevation, we add together the proportional contribution of each site type to yield the combined N mitigated each year via the denitrification process.

Figure 6. Log function between measured denitrification rate and stand age of forested wetlands.



Nitrogen losses from agricultural land are a nitrogen source to the waterway, i.e., they have a negative mitigation value, while denitrification is considered a nitrogen sink, keeping N from entering the waterway and generating a positive mitigation benefit. Since restoring a wetland on cropland precludes additional agriculture-related N losses, those forgone losses are then seen as a positive mitigation value. We assume that forgone N losses from crop production remain constant through the study period so that annual N mitigated equals the forgone N losses (37.0 kg N/ha/yr) plus the current level of denitrification. Because the agricultural site functions as the baseline, the nitrogen eliminated through denitrification there must be netted out to arrive at the N mitigated due to WRP wetlands restoration. It is assumed that the denitrification rate on cropland does not "mature" through time and so the constant mean value for the 16 CEAP agricultural sites, 1.69 kg N/ha/yr, is subtracted annually.

Figure 7. Nitrogen (N) flux accounting for MAV counties over the 90-year study period. DP is denitrification potential, WRP is the WRP sites, and Ag is the agricultural sites. Low is low elevation, High is high elevation, and Wtd Avg is 80% low, 20% high.

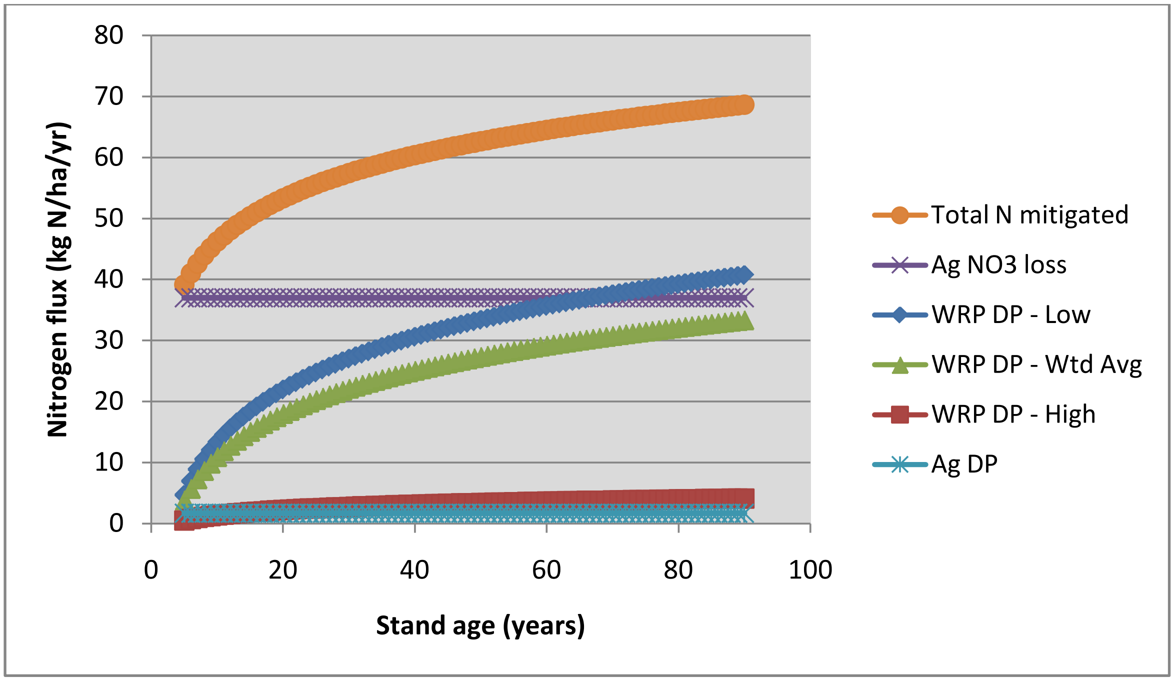


Figure 7 depicts the curves of denitrification rates for WRP low, high, weighted average, as well as for agriculture sites; the N losses associated with crop production; and the total N mitigated. Total N abatement is dominated by the cropland N loss pathway in the years immediately after a wetland restoration takes place. As the wetland grows, the contribution of denitrification to total N mitigated rises from 10% at year 5 to nearly 49% by year 90. Total N mitigated increases from about 37 kg N/ha/yr in the early years to almost 69 kg N/ha/yr by the end of the study period.

Monetizing nitrogen mitigation

Nitrogen mitigation is monetized using a price estimated for the Delta region (Arkansas, Louisiana, and Mississippi) of the U.S. South in Ribaudo et al. (2005). That study's results are selected for the benefit transfer because it is one of the few studies in the literature that produces a marginal price for nitrogen mitigation; moreover, its estimates are also specific to the MAV study area. Note that its values are only for the wastewater treatment industry.

Ribaudo et al. (2005) employ the U.S. Agricultural Sector Mathematical Programming (USMP) model to explore the potential for nitrogen credit trading in the entire Mississippi Basin by modeling the interaction between agricultural nonpoint sources and wastewater treatment plant point sources mandated to reduce nitrogen emissions. In the model, farmers are able to furnish nitrogen reduction credits via the following four methods: changing fertilizer application rates, changing production practices, growing different crops, or retiring cropland. Restoring wetlands is not included as a mitigation option because, in an earlier paper, Ribaudo et al. (2001) demonstrate that wetlands restoration is generally more expensive than fertilizer management and therefore a less attractive alternative for farmers. However, the cost of the alternative approaches does capture the avoided costs of achieving the given level of water quality improvements in another way when wetlands restoration is undertaken in the region, and thus provides a workable marginal value for wetland N mitigation outcomes.[3]

Table 4. Annualized value of N mitigation service and range of values depending on costs of marginal N credits in Ribaudo et al. (2005) (all values are in 2008 US$).

	Cost of marginal N credit ($/kg N)	Net Present Value	Annualized value ($/ha/yr)
Study area	$25.27	$30,773.76	$1,268.12
Lower bound	$22.82	$27,790.15	$1,145.17
Upper bound	$106.09	$129,196.20	$5,323.89

The cost of the marginal trade for the Delta region is estimated at $10.50/lb N, a result which we transform to $25.27/kg N by converting it to price per kilogram and then by inflating the price to 2008 dollars using the CPI Inflation Calculator (BLS 2008).[4] For the dynamic model of nitrogen mitigation developed here, the monetization step follows the same process as applied to the GHG mitigation service. Each year the amount of total nitrogen abated is multiplied by $25.27/kg N. Next, the 90-year stream of N mitigation values are discounted back to the present using a 4% discount rate and then converted to an annualized value. The result is over $1,268/ha/yr. A range of values for N mitigation is derived by using the lowest and highest N credit prices among all sub-regions in the Mississippi Basin generated by Ribaudo et al. (2005). In Table 4, the costs of a marginal N credit range from $22.82 to $106.09 kg N and the interval of annualized values is between $1,145 and $5,324. The costs to mitigate nitrogen in the MAV are clearly at the low end of the range and may therefore represent a relatively conservative estimate for the valuation of nitrogen mitigation service.

Wildlife Habitat Service

Converting row crop fields to wetlands results in additional habitat for many taxa of wildlife, including anurans (i.e., frogs), black bear, and neotropical migratory birds. Although habitat benefits accrue to a variety of wildlife in the MAV, our analysis focuses on the benefits from the expansion of migratory waterfowl habitat by WRP. This is in large part due to the widely recognized recreational value derived from waterfowl, which has generated values in the economics literature, enabling benefit transfer

[3] We recognize that replacement cost is conceptually a less-preferred shadow price than a directly estimated WTP value for the service, but unfortunately there are no direct estimates of WTP to draw from. We do believe replacement cost is an empirically valid measure for the region because policies are attempting to take a suite of approaches to achieving certain water quality targets for the region (Mississippi River/Gulf of Mexico Water Nutrient Task Force 2007).

[4] As a comparison, the Nutrient Offset Program run by North Carolina's Ecosystem Enhancement Program uses $21.67/lb N for the Tar-Pamlico River Basin and $28.35/lb N in the Neuse Basin ($47.77/kg N and $62.50/kg N) for offset payments to mitigate nitrogen (http://www.nceep.net/services/stratplan/Nutrient_Offset_Program.htm).

(Duffield and Neher 1991; Gan and Luzar 1993). Alternatively, marginal increases in anuran species or in black bear habitat have not been previously monetized.

Quantifying waterfowl habitat service flows

Flooded bottomland forests provide necessary forage for waterfowl that overwinter in the Mississippi Alluvial Valley as well as for those who stop over in the MAV en route to other wintering grounds such as Mexico (LMVJV Waterfowl Working Group 2007). Other benefits include protection from winter weather and pair isolation habitat (Baldassarre and Bolen 2006). We concentrate on the food provision aspect of these WRP wetlands which is captured by the metric Duck Energy Days (DEDs). A DED represents the amount of daily energy required by a duck supplied by a unit area of foraging habitat for a day (Reinecke and Kaminski 2007). The DED value of 294.35 kcal reflects the "average duck" wintering in the MAV, thus taking into account daily energy requirements of all dabbling ducks, of which mallards are the most common, and also of wood ducks (LMVJV Waterfowl Working Group 2007). The difference between DEDs produced on restored wetlands and on cropland is equivalent to the additional waterfowl habitat provided by WRP.

To calculate the net gain in waterfowl habitat, we first draw on the results of James et al. (in review). For the MAV areas of Arkansas, Louisiana, Mississippi, James et al. calculate the DEDs on post-restoration WRP lands, on the pre-restoration cropland, and the net DED increase for the 110-day wintering period. These calculations are based on an analysis of the flooding frequency of WRP acreage and the DED values per hectare for pertinent land use classes for WRP land (e.g., 677 DED/ha for naturally flooded restored wetland) and cropland (e.g., 89 DED/ha for harvested flooded soybean fields). In Table 5, we report the DED averages for each state over the 2001–2005 time period. The post-restoration net increase in DEDs is then divided by the total DEDs estimated to be produced in the MAV on all public and private land (LMVJV Waterfowl Working Group 2007). The quotient is the gain in DEDs in the MAV due to WRP-driven wetlands restoration, averaging 9.19% across the three states.

Table 5. Waterfowl habitat impact of wetlands conversion in duck energy days (DEDs).

State	Hectares WRP, avg 2001-2005	WRP: Post-restoration DEDs	Baseline: Pre-restoration DEDs	Net DED Increase post-restoration	Total DEDs in MAV, avg 2001-2005	DED Increase due to WRP in MAV
Arkansas	48,158	18,449,659	1,241,126	17,208,533	226,379,794	8.23%
Louisiana	65,673	10,923,441	804,859	10,118,582	132,498,674	8.27%
Mississippi	49,231	14,177,318	993,564	13,183,754	122,512,518	12.06%
Total	**163,062**	**43,550,418**	**3,039,549**	**40,510,869**	**481,390,986**	**9.19%**

The final step in this quantification process involves linking gains in waterfowl habitat to changes in hunting behavior. Increases in waterfowl habitat generally mean augmented hunting opportunities. That is, more habitat implies potentially more waterfowl in the MAV and thus a greater population to hunt. One caveat is that these waterfowl populations are migratory and thus dependent on habitat in more than one region to thrive. In particular, the prairie pothole region in the north-central U.S. and south-central

Canada serve as the most important breeding ground for North American ducks, producing 50% to 80% of the continent's duck population (Batt et al. 1989). The MAV is part of a waterfowl network called the Mississippi Flyway, whose duck populations principally originate in the prairie pothole region. Waterfowl habitat gains in the MAV represent greater resource flow in the region and create a positive network externality, though these benefits may be potentially moderated or even offset by changes in other components of the habitat network. Without modeling the entire breeding and migration network of North American ducks, our results will have to serve as a reasonable first order estimate of the region's contribution to hunting opportunity.

Greater waterfowl population numbers can result in increased harvest rates for hunters (a quality effect) as well as induce more waterfowl hunting trips (a quantity effect). More habitat provided by private land in WRP easements could also furnish additional destinations for hunting trips and thus potentially more trips (a quantity effect). We endeavor to capture these effects through a quantity measure, duck hunter days afield. A direct relationship is assumed between the percentage of increased waterfowl habitat created via WRP and the percentage increase in duck hunter days. Ideally, gains in hunter days are computed by multiplying the average numbers of duck hunter days in the MAV counties of each state for the five seasons between 2001 and 2005 by the percentage of waterfowl MAV habitat increase in the corresponding state over that same time period. Since duck hunter days are not available at the sub-state level, we use five-year averages of U.S. Fish & Wildlife Service county-level data on duck harvests to find the share of state harvest occurring in the MAV counties of the three states. These shares are then multiplied by the average number of duck hunter days in each state (2001 to 2005 seasons) to yield the number of duck hunter days in the MAV for each state (USFWS 2003; USFWS 2004; USFWS 2006). It should be noted that those percentage changes in duck hunter days, although not trivial (between 8% and 12%), are still marginal and thus appropriate for our economic valuation approach.

Table 6. The calculation of increase total surplus per hectare due to increase in waterfowl habitat in the MAV due to the Wetlands Reserve Program (WRP).

State	Increase in habitat due to WRP	Waterfowl Hunter Days in MAV, avg 2001-2005	Increase in Waterfowl Hunter Days	Total increase in consumer surplus	Consumer surplus gained per ha	Producer surplus gained per ha	Total surplus per hectare
Arkansas	8.23%	415,185	34,157	$1,655,944	$34.39	$15.00	$49.39
Louisiana	8.27%	109,383	9,044	$438,452	$6.68	$15.00	$21.68
Mississippi	12.06%	86,196	10,394	$503,910	$10.24	$15.00	$25.24
Total/Avg	**9.19%**	**610,764**	**53,595**	**$2,598,307**	**$15.93**	**$15.00**	**$32.10**

Monetizing waterfowl service flows

To monetize the change in the ecosystem service of waterfowl habitat, we consult the recreation economics literature for an appropriate value of an additional day of waterfowl hunting to be used as the transferred shadow price. For the per-day value of waterfowl hunting, we take the results of a meta-analytical study on outdoor recreation values conducted for the U.S. Forest Service (Rosenberger and

Loomis 2001). The value estimated for the southeast region was $34.72 in 1996 dollars, which we update to $48.48 in 2008 dollars by using the CPI calculator (Bureau of Labor Statistics 2008). Therefore, the total increase in consumer surplus resulting from WRP is the estimated increase in waterfowl hunter days multiplied by $48.48. Consumer surplus gained per hectare of restored wetland is simply the total increase divided by the number of hectares in WRP easements in each state. These values range from about $7/ha/yr to $34/ha/yr, with an average of $16 across the three basins. Using $15/ha/yr as the average producer surplus obtained (discussed below), that value can be added to the consumer surplus gains to yield total annual surplus values of between about $22 and $49 per hectare, with a mean of $32 across the MAV.

Total Social Value of Ecosystem Services: Partial Estimate

Summing the results from the preceding three ecosystem services valuation applications attains a partial estimate for the total ecosystem value of wetlands restoration (see Table 7). Although they were not monetized in this analysis, it is assumed that floodwater storage, sediment retention, and other habitat services also possess positive economic values. Therefore, the total social value estimated here, which ranges from $1,446/ha/yr to $1,497/ha/yr, is necessarily a lower bound on the full social value of restoring wetlands.

Table 7. Social Welfare Benefit estimates of individual ecosystem (estimates in 2008 US$/ha/yr).

Ecosystem Service	Social Value ($/ha/yr)
GHG mitigation	$162–$213
Nitrogen mitigation	$1,268
Wildlife recreation	$16
Total	**$1,446–$1,497**

As we will discuss below, the social value estimate for wetlands restoration dwarfs the market value that exists with current markets, being almost 20 times greater. However, we will first examine how it is that not all of these social welfare values can be captured in markets for the private landowner.

Market Value

The estimates in the section above are measures of social welfare value and are thus appropriate to use for social benefit-cost analysis to gauge the performance of public programs such as WRP. However, the emergence of ecosystem service markets raises the question of whether private markets can play a role in incentivizing socially beneficial landowner behavior. Thus, we turn to an assessment of market value with the potential to be captured by landowners in the region.

GHG mitigation

Market value for GHG mitigation is realized through the existence of carbon markets for GHG mitigation, wherein landowners can be compensated for sequestering carbon or reducing emissions below a baseline as part of an offset program in a cap-and-trade system. In 2008, carbon credits were traded as an environmental commodity on the voluntary Chicago Climate Exchange (CCX) in the range of $1.00 and $7.40/t$CO_2$e. We use the midpoint of this range, $4.20/t$CO_2$e, for the low market price in the analysis. Because voluntary demand is generally less binding than a mandatory system, this price is relatively small. Prices on the European Union Emissions Trading Scheme (EU ETS), part of the Kyoto Protocol compliance driven market have been much higher, near $35/t$CO_2$e in the summer of 2008, but we do not use its values because the ETS does not allow forest carbon in its trading. Instead, we draw upon the analysis of the recently proposed Lieberman-Warner climate change bill (S. 2191), which calls for a federal cap-and-trade program covering the energy, transportation, and industrial sectors with mitigation from the forest sector usable as offset credits for the capped sectors. Various estimates of the Lieberman-Warner bill estimated a carbon price of about $20/t$CO_2$e to $30/t$CO_2$e. We use $30/tonne as the upper end of the market price range. In Table 8, annualized values per hectare for GHG mitigation are calculated to be about $59 for the low market price and over $419 for the high market price.

GHG offset payments in forestry and agriculture typically have to be modified to account for permanence, additionality, and leakage (Murray et al. 2007). Permanence reflects the fact that stored carbon could be re-released due, for instance, to harvesting the timber after some time. Seeing that the majority of WRP easements in our study area are permanent, we assume that the converted wetlands will not be harvested and thus we make no adjustment for impermanence. Additionality adjusts for the fact that some of the activity getting credited may have happened anyway without the payment. This is unlikely in the case of hardwood restoration in the MAV, as afforestation rates are extremely low there without any kind of government inducement. So no further adjustment is made. Leakage means that GHG sequestration services gained in one area are partially compensated by loss in another. This can happen when restoring cropland to wetlands in one place could cause land clearing for agriculture in another. Leakage rates have been estimated at 43% for forest carbon sequestration programs in the south-central region (Murray et al. 2004). Studying 12 states in the central U.S., Wu (2000) found that about 20 acres of non-cropland was converted to cropland for every 100 acres enrolled in the Conservation Reserve Program (CRP). Nevertheless, although ecosystem service values determined here may be offset by leakage elsewhere of the system, perhaps by as much as 20% to 40%, the direct estimation of that leakage effect is outside the scope of this study. Therefore, following the protocol used by the Chicago Climate Exchange for Afforestation Offset projects(Chicago Climate Exchange 2007)., we present the calculated GHG flux values (and all other ES values estimated here) without adjusting for leakage.

Nitrogen mitigation

Although there are more than 40 nutrient trading programs on the books in the U.S., very few trades have taken place to date (Ribaudo et al. 2008). As such, the market value under existing markets is essentially zero for N mitigation. Nevertheless, given the substantial interest in nutrient trading and the degraded condition of many of the nation’s waterways, it is not implausible that N abatement will gain a market value in the near future. It should be noted that the potential market value of the nitrogen mitigation service equals only half of the social value because we assume that a nutrient trading scheme would require a trading ratio of at least 2:1. The most common ratio for trading between point and nonpoint sources is 2:1 (Morgan and Wolverton 2005). That is, two kilograms of nitrogen needs to be mitigated by farmers for every one kilogram of nitrogen credit generated. Ratios are used in order to reduce the uncertainty involved with nutrient mitigation by nonpoint sources such as farms. Therefore, we estimate an annualized potential market value of $634/ha/yr for nitrogen mitigation.

Waterfowl recreation

In addition to the consumer surplus accruing to regional waterfowl hunters, private landowners who enroll in WRP may also potentially garner some level of producer surplus. Since easements necessarily occur on private land, WRP participants can be seen as producers of the waterfowl habitat and could capture a portion of the created value through hunting leases. Recent studies in Mississippi find that hunting lease prices range from $4 to $8 per acre per season, or about $10 to $20 per hectare (Hussain et al. 2007; Rhyne and Munn 2007). Using the mean of these findings, the annual market value for waterfowl recreation is $15 per hectare.

Table 8. Benefit estimates of individual ecosystem services for market value, assuming current markets, or considering potential markets (estimates in $2008/ha/yr).

Ecosystem Service	Market Value – Current markets	Market Value – Potential markets
GHG mitigation	$59	$419
Nitrogen mitigation	$0	$634
Wildlife recreation	$15	$15
Total	**$74**	**$1,068**

Market value summary

Given current markets, market value yields about $74/ha/yr and pales in comparison to the estimated social value of over $1,400/ha/yr. However, the gap closes to a large degree when one considers potential markets for ecosystem services. At $1,068/ha/yr, the potential market value is about three-quarters of the social value and over 14 times the market value under existing markets. Nitrogen mitigation is clearly the driver for both of the larger values, comprising 59% of the potential market value and almost 90% of the social value.

COMPARISONS WITH COSTS OF WETLAND RESTORATION

To provide context for the above estimates of ecosystem service benefits, we examine the two types of costs related to their provision. The first is the private cost borne by the landowner, and the second is the social cost of implementing WRP shouldered by the federal government. We do not attempt to conduct a full cost-benefit analysis, which would imply a complete accounting of all costs and benefits of wetlands restoration. For ease of comparison with the estimated benefits, costs are converted to per-hectare units.

Landowner Perspective

From the perspective of the MAV landowner, the main opportunity cost of wetland restoration is the forgone income from agricultural use of the land. We can estimate this cost by considering either annual cash rents for agricultural land or the net returns from crop production. For the three Delta states, average cash rents per hectare range from $138 to $209, with a mean of $169 (USDA-NASS 2006). Looking at crop production in the region, returns vary substantially by crop type and by year over the period of 1997 to 2006. After subtracting operating costs from the value of production, rice emerges as the most profitable at an average of $391 per hectare, while wheat is the least at an average of $141 per hectare (USDA ERS). Using the representative agricultural hectare approach described in Nitrogen Mitigation Service subsection, we find that the annual return for a hectare of crop production in the MAV is $277.

Another relevant source of income for agricultural producers is government payment programs. The 2002 Farm Bill furnishes three types of payments to farmers, of which only the direct payment is provided annually and is independent of the crop cultivated (Ibendahl 2004). The provision of the countercyclical and loan deficiency payments hinges on national and county crop prices and is not guaranteed each year. Focusing on the Mississippi Delta, Parkhurst and Anderson (2004) calculate that the sums of the direct and maximum countercyclical payments per base acre are $17 for soybeans, $156 for rice, and $139 for cotton. The corresponding values per hectare are $42, $385, and $343. Ibendahl (2008) finds that for three Mississippi counties, expected government payments for cotton and soybeans average $133 and $25 per acre, respectively ($329 and $62 per hectare). Applying these values to the representative agricultural hectare approach, we obtain a conservative estimate of about $91 per hectare.

Using $277 as the value of a hectare for crop production and $91 as the annual government payment subsidy, their sum of $368 represents the estimated annual per-hectare income forgone by a private landowner who opts to enroll acreage in the WRP. If the landowner wished to undertake a wetlands restoration on his property without enrolling in a conservation program, one-time costs for afforestation projects in the MAV may run around $680 to $900 per hectare.[5] Assuming that those restoration costs are

[5] NRCS costs for restoring a forested wetland in Arkansas are approximately $275 per acre ($680 per hectare) (personal communication, Andrew James 2009). A private firm specializing in afforestation projects may charge around $350–$375/acre ($865–$926/hectare) for a carbon offsets package that includes the basic site preparation and tree planting, as well as "long-term carbon monitoring plan, with initial funding price inclusive of permanent monitoring plot establishment, soil carbon measurement and baseline report, 100-year carbon reporting table, and survival analysis during the third growing season," plus "guidance on offset registration and standards" (personal communication, Carol Jordan 2009).

paid up front, a present value analysis combining foregone agricultural income with the restoration costs over a 90-year horizon yields an annualized value of $400 to $411. Currently the annual market value that could be captured from existing carbon and hunting markets amounts to $74 per hectare, only about a fifth of the net returns from agricultural production. In contrast, the potential market value of GHG mitigation, nitrogen mitigation, and wildlife habitat provision with emerging ecosystem markets is $1,068—over two and a half times greater than the restoration opportunity costs. Without the payments provided by WRP, landowners will not have sufficient economic incentive to undertake wetlands restoration on their properties until markets for environmental services become more fully developed.

Taxpayer Perspective

The principal costs to taxpayers of restoring wetlands via the WRP are the easement payments to landowners and the cost share of the restoration. Easement payments provide compensation to the landowner for forgoing agriculture and are made as a lump sum in the first year of the WRP contract. Under a 30-year easement, the USDA pays for 75% of the restoration cost, whereas it covers 100% of the cost for a permanent easement (USDA-NRCS 2007). The publically available cost data for the WRP aggregates the annual costs for all three contract options at the state level for 2003 to 2007 (USDA-NRCSb). From this data, we can derive per-hectare costs incurred by the USDA for each state. The 5-year average across the three Delta states is $2,617 per hectare in 2008 dollars. Since the government no longer is obligated to provide agricultural payments when a farmer enrolls land in WRP, the annual subsidy estimated above ($91) should be subtracted from the WRP cost. We use the remainder of $2,526 per hectare as the one-time public expenditure or social cost of wetlands restoration in the MAV.

Again considering the values reported in Table 8, it would only take two years for the social benefits of wetlands restoration (~$1800/ha/yr) to surpass the costs incurred by the government in paying for the WRP. Furthermore, the estimated social benefits represent a lower bound on the total ecosystem value since several ecosystem services are not accounted for in the analysis. The ecosystem service value return on public investment appears to be very attractive in the case of the WRP.

BENEFIT AGGREGATION FOR MAV

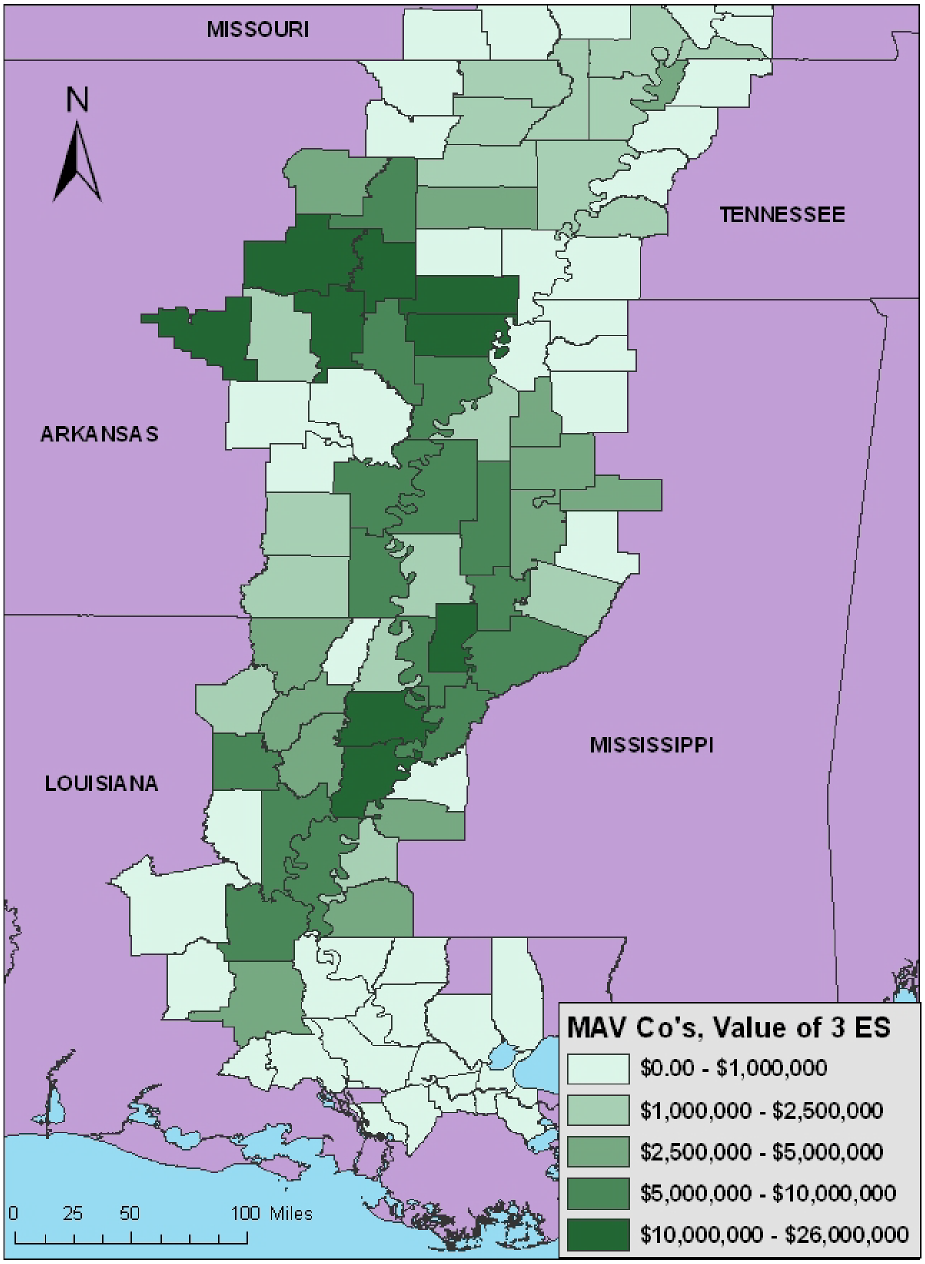
The measurement of aggregate benefits resulting from a program can be useful to policymakers by providing an estimate of the magnitude of program impacts. Using the per-hectare values for the three focal ecosystem services, we can scale them up to generate aggregate values for the study area, the three major river basins of the MAV. Examining the benefits associated with the land currently enrolled in WRP there, we observe that there are 226,522 hectares in WRP easements in the 104 counties in the MAV (as of 2005). With the assumption that the services are provided equally by each WRP hectare, we apply their social welfare values, which are \$213.40 per hectare for the GHG mitigation value (using \$15/tCO_2e), \$1268.12 for the nitrogen mitigation, and \$15.93 for waterfowl recreation. Multiplying these values by the number of WRP hectares located in each county, we calculate county-level estimates of the bundled values of the three services and then sum those to arrive at an aggregate value at the spatial scale of the MAV (see Table 9).

Table 9. Annual GHG mitigation, nitrogen mitigation, and waterfowl recreation values (2008 US\$) for WRP land combined at the MAV level.

	Extent (ha)	GHG mitigation	N mitigation	Waterfowl recreation	Aggregate value
WRP per hectare		\$213	\$1,268	\$16	\$1,497
All WRP land	226,522	\$48,339,795	\$287,257,079	\$3,608,495	\$339,205,369

The differential distributions of bundled ecosystem service values across the study area counties is reflected in Figure 8, a map displaying the value of the three ecosystem services on WRP land for each of the counties. Higher values are represented by progressively darker shades of green coloring the counties. Annual MAV-level benefits are approximately \$339 million, although 25 of the 104 counties supply almost 75% of the value.

Figure 8. Counties of the Mississippi Alluvial Valley (MAV) by annual aggregate social value of the three bundled ecosystem services generated on restored wetlands on WRP land.



CONCLUSIONS

As public goods, ecosystem services are underprovided because they are undervalued in the marketplace. Thus far, government programs such as WRP and CRP have sought to increase the flow of these services, and they have attained a certain level of success, as has been demonstrated by this analysis. However, with increasing public recognition of the importance of healthy ecosystems to human welfare also comes the potential for new economic opportunities in the form of private ecosystem markets. Policymakers and business entrepreneurs need good information on the economic value of ecosystem services to guide their programs and market development efforts. This paper addresses that need.

The Mississippi Alluvial Valley is a particularly rich ecosystem that has undergone massive change in the last 100 years. It has been a recent target of restoration efforts through WRP, CRP, and other programs. To examine ecosystem service values from WRP restoration in this region, we combined field data collection with secondary data collection and then linked these data with process models to calibrate expected change in those values. Unlike many other ecosystem service studies that have used top-down, landscape-level approaches, we implemented a bottom-up integration of ecosystem service function measurements, environmental modeling, and economic valuation.

Focusing on three services—GHG mitigation, nitrogen mitigation, and waterfowl habitat—we estimated a lower bound for the economic value to society of restoring wetlands in the MAV. With advances in methodologies and markets, that value will likely grow as currently unmonetized services, such as floodwater storage, gain their own price tags. Considering the lower bound estimate, this study’s findings suggest that restoring wetlands in MAV has a total economic value to society well above the alternative use in agriculture. The largest benefits are found to flow from nitrogen mitigation, followed by GHG mitigation. Nevertheless, absent expanded public programs or new ecosystem service markets to deliver payments, landowners are being economically rational by keeping most of this land in agriculture, which currently has a higher market return. As a result, some mix of expanded payments from the public or private sector would appear to be warranted to incentivize continued wetlands restoration at a net benefit to society.

From the taxpayer perspective, the social benefits easily outstrip the social costs of restoring wetlands via WRP, as the public investment pays for itself in enhanced ecosystem services in only two years. Again, these benefit estimates do not include other services that do not presently have a clear monetary value, but may in the future. Given the considerable “surplus” in conservation effects generated by WRP payments, there could be substantial opportunity for mitigation markets in the region to supplement, or possibly even replace, conservation program payments.

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