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# Targeting Investments To Cost Effectively Restore and Protect Wetland Ecosystems: Some Economic Insights

LeRoy Hansen, Daniel Hellerstein, Marc Ribaudo, James Williamson, David Nulph, Charles Loesch, and William Crumpton

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## Abstract

USDA has spent more than $4.2 billion on wetland restoration and protection over the last two decades. One challenge in allocating these funds is the lack of information on variations in wetland benefits and costs across the Nation. This report discusses the biophysical impacts of new wetlands for eight benefit categories: duck hunting, carbon sequestration, flood protection, nitrogen removal, species protection, open space, sediment removal, and groundwater recharge, as well as the value of these impacts for some categories. In addition, it presents county-level estimates of the costs of restoring and preserving wetlands for some parts of the United States. Although the estimates range in precision and are not comprehensive, they call attention to some areas where the benefits of new wetlands are likely to exceed costs or perhaps may be insignificant. For example, the benefits of restoring and preserving wetlands near the Missouri River in central North and South Dakota are likely to exceed costs. Findings underscore the need for additional information that may increase the number, accuracy, and spatial resolution of wetland benefit estimates.

**Keywords:** Wetland conservation, ecosystem benefits, environmental economics, environmental targeting, nonmarket benefits, values of wetland ecosystems

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# Targeting Investments To Cost Effectively Restore and Protect Wetland Ecosystems: Some Economic Insights

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## What Is the Issue?

Over the last several decades, concerns over the loss of wetland ecosystems have led to legislation restricting wetland conversion and promotion and funding of wetland restoration and conservation. Consequently, USDA has spent more than $4.2 billion on wetland restoration and protection during the period. The environmental benefits of wetland conservation—cleaner water, increases in wildlife populations, and carbon sequestration—depend on climate, human influences, and landscape characteristics. This study assesses how future wetland conservation funding might be targeted within States and regions and across the United States to maximize benefits relative to costs of restoring and preserving wetlands. Information on the sizes and spatial distributions of the benefits and costs of wetland protection can help public and private decisionmakers understand the economic implications of efforts to reduce wetland losses, improve existing wetlands, and restore prior-existing wetlands through regulation and economic incentives.

## What Did the Study Find?

A number of different types of wetland benefits were considered as part of this analysis. The ranges in wetland benefits and costs across the country demonstrate potential gains from targeting public and private wetland restoration efforts by location and by wetland characteristics. Some benefits can be estimated more precisely than others:

- The annual benefits to duck hunting from restoring wetlands in the Prairie Pothole Region (primarily, lands in the eastern Dakotas and western edges of Minnesota and Iowa) range from near $0 to $143 per wetland acre.
- The annualized value of a new wetland's effect on atmospheric concentrations of greenhouse gases ranges from $0 to $129 per wetland acre in the areas studied. In other areas, wetlands can increase, decrease, or have no net effect on atmospheric levels of greenhouse gases.
- Wetlands on agricultural land are not likely to provide notable flood-protection benefits unless they are located in close proximity to urban areas.
- Wetlands in the Upper Mississippi and Ohio River watersheds can annually remove from 11 to 1,800 pounds of nitrogen per wetland acre in waters leaving farm fields from watershed streams and rivers.

ERS is a primary source of economic research and analysis from the U.S. Department of Agriculture, providing timely information on economic and policy issues related to agriculture, food, the environment, and rural America.

- About 1,756 counties in the contiguous States have wetland-associated imperiled species, with some counties having as many as 23 imperiled species. About 1,355 counties have no wetland-associated imperiled species, but this does not suggest that the value of wetland-species protection benefits in these counties is low. The analysis found no information that illustrates how a new wetland might affect species' survival probabilities. Additionally, it found no information on the public's willingness to pay (or how it might prioritize support) for changes in imperiled species' survival probabilities.
- Many wetlands recharge groundwater supplies, but the majority do not. Some wetlands exist because they are fed by groundwater. Others exist because they have impermeable subsoil.
- Sediment removed by wetlands, while protecting other bodies of water, can reduce other wetland benefits as the sediment accumulates.
- The benefits of wetlands as open-space protection in rural areas are likely to be very small.

Based on contract data from USDA's Wetlands Reserve Program, the costs of restoring and preserving a wetland range from $170 per acre in the western Dakotas, Montana, Arkansas, and Louisiana to $6,100 per acre in the major corn-producing areas and along the Northern Pacific Coast. These estimates reflect the change in the value of the land with and without a permanent wetland easement plus wetland restoration costs.

An analysis of the combined benefits and costs of new wetlands revealed the following:

- Restoring wetlands in the western Prairie Pothole Region is likely to generate duck-hunting benefit-cost ratios close to or greater than one; that is, restoration funds spent in this area would be a good investment. However, most of the people who would benefit from the added duck populations in the region are hunters who live elsewhere.
- The value of new wetlands' greenhouse gas impacts exceeds wetland costs in the western half of the Prairie Pothole Region and in most of the Mississippi Alluvial Valley (from southern Illinois, covering western portions of Tennessee and Mississippi and eastern portions of Missouri and Louisiana).
- The amount of nitrogen that can be removed for each dollar spent on wetland restoration and protection in the Upper Mississippi and Ohio River watersheds ranges from 0.2 to 34.0 pounds. The costs of nitrogen removal through new wetlands range from $0.03 to over $5.00 per pound of reduced N loadings in nearby waterways.

## How Was the Study Conducted?

To quantify wetland ecosystem effects on amenities, and the value the public places on these amenities, ERS researchers reviewed and analyzed economic, ecological, and engineering literature and data. In some cases, researchers used existing data and models; new models were also developed. Benefits were estimated at the county and subcounty levels across limited portions of the Nation, depending on the spatial resolution (e.g., geographic reliability) of supporting data and economic and ecological models.

Researchers used Wetlands Reserve Program contract data to empirically estimate wetland cost models for major wetland regions. These models were then used to generate county-level estimates of the costs of new wetlands. Where the estimates allowed, researchers generated county- and subcounty-level benefit-cost ratios by dividing the benefit estimates by the cost estimates. These ratios illustrate areas where wetland restoration funding might be allocated to best serve public interests. When it was not possible to estimate economic benefits, researchers generated ratios of the wetlands' cost effectiveness—the quantity of services produced relative to costs. Lastly, where data were limited, representative measures were generated that illustrate the scope and the magnitude of possible benefits.

# Targeting Investments To Cost Effectively Restore and Protect Wetland Ecosystems: Some Economic Insights

## Introduction

Since European settlement, the 48 contiguous States are estimated to have lost about half of the estimated 220 million original wetland acres. Some areas lost greater shares than others—regions within Iowa, Illinois, and Indiana lost as much as 90 percent. Most losses stemmed from drainage for agriculture, as farmers found that the land provided returns that covered the cost of wetland conversion. In fact, prior to the 1970s, the Federal Government provided subsidies for some wetland conversion.

In more recent years, the breadth of ecological services provided by wetlands, ranging from serving as critical habitat of many plant, fish, and wildlife species to carbon sequestration, has been widely recognized and appreciated. Nonetheless, economic incentives for private landowners to provide most wetland services are generally weak—the benefits provided by wetlands and their ecosystems often have a nonmarket "public goods" nature. Furthermore, the individuals that benefit may be far from the wetland, making the collection of monetary returns all the more difficult.

Public goods are characterized as being nonrival in consumption (an additional individual can enjoy the good without affecting others) and by the difficulty of excluding nonpayers (Kaul et al., 1999). For example, the view of a wetland from public land, such as a highway, is a pure public good. Similarly, wetlands can boost duck populations. But it is difficult to exclude those who are not willing to pay for the subsequent increase in hunting quality and views of migratory ducks. Generally speaking, even though some amenities provided by wetlands are not pure public goods, the public-good characteristics (as well as wetlands' effects on other amenities) are likely to make market provision inefficient (Ribaudo et al., 2008).

Since the 1970s, having recognized the public goods provided by wetlands, Federal, State, and local governments, as well as nongovernmental organizations (NGOs), have initiated efforts to reduce wetland losses, improve existing wetlands, and restore prior-existing wetlands through regulation and economic incentives. USDA's most significant wetland restoration effort was the Wetlands Reserve Program (WRP), a voluntary program established by the Food, Agriculture, Conservation, and Trade Act of 1990 (the 1990 Farm Act).[1]

[1]The Agricultural Act of 2014 rolled functions of the Wetlands Reserve Program, Grassland Reserve Program (easement portion), and the Farm and Ranch Lands Protection Program into the Agricultural Conservation Easement Program (ACEP). All land enrolled in these earlier programs is now considered enrolled in ACEP. The three earlier programs received average annual funding of $686 million for 2008-13, but annual ACEP funding for 2014-18 is expected to average $375 million.

The objective of the WRP was to protect, restore, and enhance wetland ecosystems to protect and improve:

- Habitat for migratory waterfowl and other wetland-associated wildlife, including imperiled species
- Water quality and groundwater recharge
- Floodplains and water flows to reduce flooding
- Aesthetics and open space, and native flora and fauna
- Education and scientific scholarship

By the end of 2012, approximately 2.6 million acres were enrolled in the WRP, of which 2.1 million acres are restored wetlands, about 0.14 million acres are prior-existing wetlands, and the remainder are surrounding uplands.[2] The WRP's financial commitments varied over time but averaged about $200 million per year from 2009 to 2012.

The Conservation Reserve Program (CRP) is USDA's other major wetland conservation effort, though wetland conservation is not the primary goal of the program. By the end of 2012, the CRP had 2.29 million wetland-related acres under contract, including 0.50 million acres of improved, prior existing wetlands; 0.54 million acres of newly restored wetlands; and 1.251 million upland acres (e.g., dry lands surrounding wetlands). In 2012, annual expenditures on CRP wetland-related contracts totaled about $175 million.[3] Together, the WRP and CRP protect about 2.6 million restored and 6.4 million prior-existing wetland acres.[4]

Four features in the design of the WRP and CRP are noteworthy, particularly in regards to wetland conservation programs and NGO initiatives. First, for both the WRP and CRP, a participating land-owner maintains ownership of the land, is allowed limited uses of the land, and can sell the land. However, the contractual agreement, and its associated constraints, stays with the land in the event of a transfer of ownership.

Second, the lengths of the contracts vary by program. About 87 percent of the WRP contract acres are permanent easements, 6 percent are 30-year easements, and the remainder are 10-year agreements. In contrast, CRP wetland-related contracts range from 10 to 15 years (about 25 percent are for 10 years, 75 percent are for 14 or 15 years). The differences in contract lengths may affect the types of benefits generated. For example, some wetland ecosystems take years to recover. In such cases, the annual services of newly restored or improved wetlands might increase over time as they move toward their full potential.

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[2]Total WRP acreage data are from www.nrcs.usda.gov/Internet/NRCS_RCA/reports/fb08_cp_wrp.html. The breakdown of Wetlands Reserve Program (WRP) contract acreages is based on ERS analysis of detailed data on WRP contracts supplied by USDA's Natural Resources Conservation Service. The detailed contract data file has a little more than half of all WRP contracts. We have assumed that the distribution of contract types in it is representative of all WRP contracts.

[3]Estimates are based on CRP data supplied by USDA's Farm Service Agency and include only yearly rental payments (cost share expenses for installing land covers, and the cost of signup incentive payments, are not included).

[4]At the end of fiscal year 2014, the CRP had 2 million wetland-related acres under contract (http://www.fsa.usda.gov/Internet/FSA_File/summarysept2014.pdf). At the time of publication of this report, WRP contract data were not available beyond 2012.

Third, the program payment schedules vary by program. In the WRP, a landowner receives a one-time upfront payment from the Federal Government. In contrast, CRP contracts are agreements under which the landowner receives annual payments. This design feature engenders comparisons of benefits and costs provided by the two programs over time. The WRP is an investment that provides wetland benefits, in most cases indefinitely, whereas the annual CRP payments are more analogous to annual rental payments, and the stream of environmental benefits flowing from the retired land may decline after the contract expires. From 2009 to 2013, approximately 50 percent of the CRP acres in expiring contracts were re-enrolled.[5] For the other 50 percent, historical use suggests that a portion of the land might return to agricultural production—these lands were typically in crop production in the 1980s and, while profitable then, recent increases in commodity prices have boosted returns. But under the following circumstances, landowners may choose not to drain wetlands: the land is more profitable if left as a wetland (i.e., it may be profitable to allow forested wetlands to mature); the landowner enjoys having a wetland; or the costs to covert the land are prohibitively high. And, if wetlands are left undrained, wetland ecosystems will continue to produce public benefits without public funding.

Lastly, competitive enrollments can reduce program costs and increase benefits. Generally, lands that can provide wetland benefits are accepted in the CRP as long as they meet certain baseline requirements, such as having been used in crop or livestock production.[6] Enrollment in the WRP is competitive. To participate, landowners submit offers to USDA's Natural Resources Conservation Service (NRCS) State field offices. The State field offices assign scores to each offer based on ranking criteria. Most State offices develop their own ranking criteria, though some States have worked together. The ranking criteria award points to contract offers based on a range of expected environmental effects and, in some States, costs. Within each State, offers receiving the highest scores are generally accepted, given budget constraints and acreage enrollment targets.

Federal, State, and local governments and NGOs that wish to develop and improve ranking criteria that represent public interests in wetlands' environmental impacts face at least two major challenges. First, they must determine how a new wetland and its ecosystem might affect environmental amenities. Second, they must assign weights to the various wetland amenities that reflect the relative strengths of public preferences. Because the information needed to make these determinations is limited, ranking criteria are based on the judgments of program managers, individuals, community groups, NGOs, and commercial enterprises.

This study assesses how future wetland conservation funding might be targeted within States, regions, and across the United States to maximize benefits relative to costs. It uses data, model results, and findings from previous research to generate spatial estimates of the costs and benefits of new wetlands. That is, to the extent possible, the analysis taps biophysical and economic sciences to quantify wetlands' and their ecosystems' effects on amenities, the public's preferences for amenities, and wetland costs. Findings offer geographically disaggregate illustrations of costs and some environmental benefits and impacts in some regions of the United States. Findings also illustrate the many areas where more information is needed.

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[5]This result is based on an analysis of all CRP contracts. Current data limitations do not allow for an evaluation of expiring wetland-related CRP contracts.

[6]The general CRP uses a competitive auction mechanism to select acres. However, most CRP wetland contracts are enrolled via the continuous CRP, which accepts any offer that meets eligibility requirements.

# The Costs of Wetland Establishment

The primary data used in the cost analyses come from WRP contracts. The WRP file includes the amounts of the payments accepted by landowners for the easement and wetland restoration—the dependent variable in the cost functions. The WRP contract data are particularly appealing because the program, by congressional design, caps easement payments at the difference between the values of land before and after it is enrolled. When administered correctly, easement payments are not likely to overcompensate landowners and are a good measure of opportunity costs. WRP data do not include the cost of technical support provided by USDA. Thus, the cost estimates may understate the total cost of restoring and protecting a wetland. Any bias in the estimates, however, is expected to be relatively small.

Clearly, larger wetland easements will require greater easement (incentive) payments and have greater restoration costs, all else being equal. But a number of other factors affect the size of the payments landowners are willing to accept and the wetland restoration costs. The WRP data provide other variables, including a county identifier, which enable one to link each observation to other county-level data.

## Drivers of wetland easement and restoration costs

Restoring and preserving a wetland through a publicly funded voluntary program involves two types of costs: the net opportunity cost (current minus the alternative land use values) is the cost of the permanent wetland easement;[7] and the cost of restoring wetland hydrology can include simple measures (blocking or slowing water movement) or significant construction (dredging wetland fill), as well as the establishment of appropriate flora and wildlife habitat in the wetland.

Given the program's features and based on a review of the literature, the biggest drivers of wetland *easement* costs are the following:

1. *Current land value*—Landowners will want to be compensated for what they give up, which will usually be less than the market value of their land, given that the remaining property rights have value. The current market value of farmland is at least as great as its agricultural value but can be higher when there is a potential for urban or other higher valued uses.

2. *The value of the land with an easement*—The amount that a landowner gives up when selling the easement (e.g., the opportunity cost of the easement) also depends on the value of the land with the easement—the compensation payment a landowner demands will be inversely related to the value of the land with the easement. The market value of land with an easement will be at least equal to its remaining agricultural-use value (i.e., occasional grazing and haying) plus the value of the land for onsite recreation (i.e., bird watching and hunting), but it could be greater. It is expected that the remaining agricultural value will be correlated with the size of the easement and, possibly, its pre-wetland agricultural value. The onsite recreation value is expected to be directly affected by the number of people likely to use the wetland, which is likely to be closely related to the size of the surrounding population.[8]

[7]Past research on wetland benefits has failed to recognize the value of the land with wetlands. See, for example, Jenkins et al. (2010) and van Kooten et al. (2011).

[8]Studies provide strong evidence that, in the past, demand for lands with wetlands can sometimes drive the value of the lands above their agricultural and alternative-use values (Reynolds and Regalado, 2002: Earnhart, 2001; Pease et al., 1997).

3. *The value of land near the wetland*—Landowners who own other land near the wetland are likely to consider the effects of the wetland on the values of those lands—should the value of surrounding lands increase (decrease), the landowner is likely to accept a lower (higher) easement payment. Findings suggest that the value of land surrounding a wetland easement can increase when buyers are willing to pay a premium for land with a wetland view (Ma and Swinton, 2011; Mahan et al., 2000; Doss and Taff, 1996)—the greater the urban pressure, the greater the premium. Note that Reynolds and Regalado (2002) found that the type of wetland affects the premium. But findings also suggest that a restored wetland nearby can reduce the urban development value of land. In an analysis of the rapidly growing Raleigh-Durham-Chapel Hill region of North Carolina, Kazak and BenDor (2013) found that a new wetland lowered the urban-development value of land within 0.5 miles of the wetland. While urban development pressure can have some effect, the direction of the effect, if any, is uncertain.

4. *Nuisance cost*—Nuisance cost is the increase in fuel, time, and fencing needed to maneuver equipment around and keep livestock out of new wetlands—or, alternatively, the opportunity cost of altering farm production or cattle grazing on the surrounding lands. The nuisance cost is borne by the landowner. Easement payments will need to offset nuisance costs to encourage landowner participation (Pattison et al., 2011). It is expected that larger contracts will impose greater nuisance costs, driving up the cost of an easement.

5. *Good steward*—A landowner wishing to be a "good steward" may accept a lower easement payment. While good stewardship can have an effect on the easement payment a landowner would find acceptable, this analysis does not include an explicit measure of such.

The primary drivers of (or factors affecting) wetland *restoration* costs are the following:

1. *Size*—Larger easements require more restoration resources.

2. *Hydrology*—The cost of restoring a wetland's hydrology depends on the level of construction needed. For example, filling channels, eliminating tile drainage, and restoring flow impediments can be much less costly than excavating filled wetlands. The actions taken to restore wetlands' hydrology would not be expected to differ substantially across wetlands of the same type.

3. *Flora and fauna*—The cost of restoring flora and fauna that support wetland ecosystems depends on the type of activities involved, such as planting seeds or planting live vegetation, transporting live fauna or allowing natural recovery, planting by machine or by hand, and preventing or controlling invasive species. For example, restoring grassed wetland may only require drill-seeding the land prior to restoring the hydrology, while restoring a forested wetland may require planting young plants by hand and, hence, may be very labor intensive. The actions taken to restore wetlands' fauna would not be expected to differ substantially across wetlands of the same type.

Based on these drivers, a database was developed to serve as the explanatory variables (including variable proxies) in a model of wetland costs. We first considered current farmland values. Because spatial data on farmland values nationwide are not available, we used county-level data on farmland rental rates—which are, effectively, an annualized measure of the agricultural value of land. We therefore generated a proxy variable for the agricultural value of land in an easement by multiplying the rental rate values by the number of acres in the easement. Note that, as discussed earlier, the agricultural value of land is expected to have positive effects on lands with and without wetland easements.

Another variable included in the database serves as an indicator of urban pressure. Urban pressure can drive up farmland values—the greater the potential for nonfarm development, the greater the land value. But urban pressure can also drive up the values of lands with easements and lands surrounding easements. Hence, the net effect of urban pressure is uncertain.

A third variable included is the size of the wetland. For a given type of wetland, the cost of restoring the wetlands' hydrology and ecosystems and nuisance costs are expected to increase with the size of the wetland. The agricultural value of land with a wetland is also expected to increase with easement size, which may partially offset the other two effects. However, there may be economies of scale or fixed costs associated with restoration and the other factors so that costs may not be proportional to the size of the wetland. To account for nonlinear relationships, a higher order term of this variable was included in the regression models.

To control for differences in restoration costs across wetland types, separate models were estimated for each of 10 major U.S. wetland regions, as defined by NRCS, where adequate data are available (fig. 1). By design, these regions reflect wetland similarities. It seems reasonable to assume that, within a wetland region, determinants of the cost of restoring a wetland—other than the size of the wetland—are likely to be similar. The demand for the view of a wetland will depend on the appearance of the wetland and the type of ecosystem it supports, which is expected to show little spatial

Figure 1
**Major wetland regions of the contiguous States**


USDA, Natural Resources Conservation Service.

variation for a given type of wetland. Regional analysis helped control for several drivers of the cost of wetland easements. (For details on the model variables, see appendix 1.[9])

## Predicting the public cost of establishing new wetlands

Across the 10 models, most coefficients are statistically significant at the 99-percent level, and the estimated models explain 53 to 88 percent of the variation in wetland costs (as measured by each model's adjusted R-square). The five best-fitting wetlands cost models capture more than 75 percent of the variation in total wetland cost, and the three weakest capture 53 to 60 percent. (For estimation results, see appendix 1.)

Using these results to generate county-level estimates of the total cost of a new wetland requires selecting appropriately representative sizes for each wetland region. For this analysis, we chose the median size of previously established WRP contracts (within a region), as this facilitates generating an estimate of median cost. Given the nonlinearity of the cost models, the average-sized contract will not generate an estimate of the average contract cost. We also used county-specific measures of agricultural land rental rates. Thus, the county estimates are driven by a regionally representative median contract size, the region-specific estimated coefficients, and county-specific agricultural land rental value.

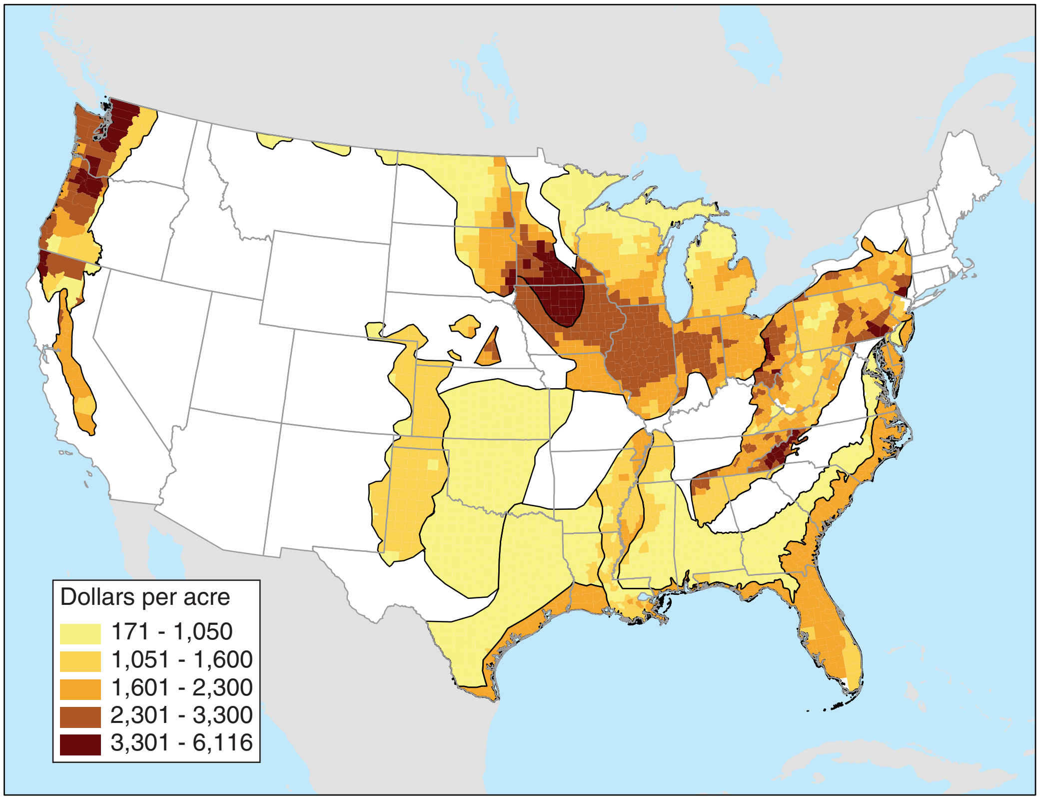
To allow cross-region comparisons, we converted total wetland cost estimates to per-acre costs. The per-acre cost estimates range from $170 to $6,100, with some of the lowest costs in western North Dakota and eastern Montana and the highest in major corn-producing areas and western Washington and Oregon (fig. 2).[10] Wetland costs are most variable spatially within the Prairie Pothole Region (where values increase from northwestern to southeastern extremes), Northwest (where values are high in much of the region), and Appalachian Plains (where values vary sharply throughout). Wetland costs could not be estimated for areas that lie outside the NRCS wetland regions examined in this analysis.

[9]We included a variable that was meant to account for the effects of urban proximity, found it to be statistically insignificant, and dropped it from the final model. Our belief is that urban proximity has offsetting effects. The potential for urban development can boost farmland prices above their agricultural values (which will tend to increase the cost of an easement). Conversely, urban proximity can increase the values of land and surrounding wetlands because of greater demand for wetland amenities (see appendix 1).

[10]Lawley and Towe (2014) recently examined the cost of easements on existing wetlands that cap land uses at the current levels in the Prairie Pothole Region (PPR) of Canada. They found that easement payments averaged $86 per acre (U.S. $). Two primary factors suggest that, although low, the Lawley and Towe (2014) estimate is consistent with the ERS estimate in this study. First, the easements in this analysis are placed on land that had been drained and is being cropped, while Lawley and Towe (2014) evaluated easements on existing wetlands (e.g., land in a lower valued use). Second, the estimated pattern of easement costs (they drop as one moves northwesterly) in this study suggests that easement costs would be lower in Canada (in an area that is north and west of the U.S. PPR).

Figure 2

**County-level estimates of the cost of restoring and preserving a new wetland**




White denotes areas where cost estimates were not calculated. The major boundary lands on the map define the wetland regions shown in figure 1.
Source: USDA, Economic Research Service using data from Wetlands Reserve Program contract files.

# Selected Estimates of the Benefits of Wetland Conservation

Wetlands have been credited with increasing and enhancing water quality, protecting fish and wildlife populations, providing flood protection, sequestering carbon, and providing open space and aesthetic beauty.[11] This section briefly describes the biophysical and economic models used to estimate wetland services, amenities, and the value of amenities.[12] We then present a summary of findings on the values of a suite of eight types of wetland benefits within a number of wetland regions. The estimates of wetland benefits are reported in annual values. To provide a more intuitive comparison of costs and benefits, we convert wetland costs to annualized values using a 4-percent discount rate.[13]

The eight wetland benefits examined include the following:

- Duck hunting
- Nitrogen removal
- Species protection
- Flood protection
- Greenhouse gas reductions
- Groundwater recharge
- Sediment removal
- Open space

The choice of these eight benefits is driven by the availability of data, the ability to develop biophysical and economic models, and findings of past research that have some level of spatial resolution. This group reflects some of the often-stated benefits of wetlands. Wetland benefits not addressed here include impacts on other wildlife, increased biodiversity, reduced phosphorous and pesticide loadings, and improved recreational and commercial fisheries. We were unable to generate most of the benefits across all regions. The results presented here do not reveal the optimal allocation of U.S. wetland restoration funds because the number of benefits estimated is limited and confined to only parts of the United States. Even so, the estimates (1) provide a perspective on the size and spatial variation in some benefits in some areas; (2) can aid judgments on how to weave reported and unreported economic and biophysical information together to support information-based decisions on targeting wetland conservation funding; and (3) illustrate the need for additional data and analyses to further support information-based decisionmaking.

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[11]For example, see the U.S. Environmental Protection Agency's discussion on wetland impacts at: http://water.epa.gov/type/wetlands/outreach/upload/EconomicBenefits.pdf

[12]Not all wetland "benefits" are positive—some people may look at wetlands as a source of mosquitos and displeasing odor.

[13]We apply a 4-percent market rate of discount in this analysis, the longrun return on AAA bonds (U.S. Federal Reserve, 2012), because interest rates observed on AAA bonds are reflective of the time preference for money, void of any risk premium.

## Estimating the value of wetlands and their ecosystem services

To estimate the value of a new wetland's benefits, one must apply economic and biophysical sciences. Biophysical sciences enable one to quantify the services that a new wetland and its ecosystems are likely to produce. The same sciences can link changes in services to changes in environmental amenities—that is, the things people care about. Economic analyses can enable one to estimate the public's willingness to pay for changes in amenities.[14]

Each of these economic and biophysical tasks is critical; the strength of a wetland benefit estimate is only as strong as its weakest link. As one's understanding of each link improves, so, too, will the analyses of wetlands' spatial benefits.[15] Furthermore, a better understanding of these links will help one better understand the tradeoffs embedded in wetland ranking criteria.

Table 1 lists some of the services wetlands and their ecosystems produce, the amenities that services produce, and the ways in which the public values amenities. Appendix 2 provides details on the biophysical and economic framework critical in evaluating the benefits of a new wetland.

This framework for estimating the value of benefits requires several kinds of information (both data and models). The specific units that define a wetland service and the amenities provided by each service are "grey" in that the way in which they are specified by an analyst will depend on the available data, models, and other information. What is important is the conceptual framework that the functions illustrate. When this information is not fully available, a number of shortcuts can be taken, including reduced-form models, proxy variables, and meta analysis. But, as discussed in appendix 2, several features leave meta analyses unsuited to generate spatially disaggregated estimates of benefits.

## Duck hunting benefits: What the study found

Wetlands directly affect duck populations. Between 55 and 80 percent of all North American ducks nest in the Prairie Pothole Region (PPR) (see fig. 1). Ducks of the PPR region migrate using all of the continent's major flyways, including the coastal flyways. Increases in duck populations generate a variety of amenities that people throughout the country enjoy:

- Improved hunting quality
- Better bird watching throughout the year
- An increase in the size and number of fall migrating flocks

This study focuses on wetlands' impacts on hunting quality and people's willingness to pay for the changes.

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[14]Valuing changes in amenities is complicated by the nonmarket nature of these goods. Instead of market prices, valuation models are used to calculate a marginal price. (See box "Analytical Methods for Valuing Environmental Amenities" for a short list and brief description of methods used to value environmental amenities.)

[15]Knowledge of these links is also necessary when evaluating benefits of on-field conservation incentives. Additionally, analysts face the challenge of quantifying indirect effects of the incentive payments, where indirect effects are defined as the changes in farming practices made elsewhere in response to the conservation incentive (Smith and Weinberg, 2004). For example, incentives that discourage some farmers from sequential years of corn production can increase the price of corn and increase corn production (and subsequent environmental impacts) elsewhere.

Table 1

**Functional relationships between new wetlands' direct effect on the environment and the value the public places on the subsequent impacts on amenities**

| Biophysical and environmental processes | | | Economic valuations |
|---|---|---|---|
| *Resource* | *The services produced by wetland x* | *Amenities that stem from services* | *The public's demands or willingness to pay for amenities* |
| | $f_i(x) \rightarrow$ | $g_j(f_i(x)) \rightarrow$ | $D_k(g_j(f_i(x)))$ |
| ECOSYSTEM SERVICES | Habitat for terrestrial and aquatic species | Improved species' survival rates | What are people willing to pay to better protect species? |
| | | Improved duck hunting quality | What are people willing to pay for better hunting? |
| | | Increased health of game fish species | What are people willing to pay for better recreational fishing? |
| | | Increased health of commercial fish species | What is the change in the commercial fishing industry's profits? |
| | Nutrient removal | Less algae in freshwaters | What are people willing to pay to have cleaner water? |
| | | Commercial fish populations less affected by hypoxic zones | What is the change in the commercial fishing industry's profits? |
| | | Lower nitrogen level in waters used by municipalities | What are the municipal water-treatment costs savings associated with reduced nitrogen in intake waters? |
| | Reductions in greenhouse gas | Economic, ecological, and health impacts of climatic change | What are people willing to pay for a reduction in the economic, ecological, and health impacts of climatic change? |
| SERVICES | Groundwater recharge | Drinking water supply | What is a municipality willing to pay for an increase in water supply? |
| | | Groundwater use in agriculture | How much will increases in groundwater affect agricultural profits? |
| | | Groundwater's effect on stream amenities | What are people willing to pay to see an increase in the amenities? |
| | Floodwater retention | Reduced flood heights (wetland bounce) and water speeds | What is the expected value of the reduced expected property damages? |
| | | | What are people willing to pay to reduce risks to health and safety? |
| | Open space | Improved views | What are people willing to pay to view a wetland, given its location? |

Source: USDA, Economic Research Service.

## Analytical Methods for Valuing Environmental Amenities

The following descriptions are meant to strengthen our illustration of the analytic framework supporting the valuation of wetland benefits. For more details on these and additional approaches and their application to wetland benefit analyses, see Brander et al. (2006).

**Contingent Valuation** relies on surveys of a sample of the relevant population. Individuals are asked about their willingness to pay for (or accept) a change in an environmental amenity. There are many variations of this approach, but they involve surveys on preferences. The collected data are then used to estimate the public's demand (or willingness to pay) for the amenity (Loomis and Walsh, 1997).

**Travel Cost Method** uses survey data that included details on the quality of the amenity each survey respondent sees, personal information, travel time and costs, and recreation activities associated with the amenity. Changes in consumer surplus associated with a change in the amenity are derived from the estimated demand function.

**Hedonic Analyses** derive social values of environmental amenities by estimating demands for market goods as a function of (among other things) the quantity and quality of the amenities. This approach is suitable when the existence of an amenity directly affects the price individuals are willing to pay for a market good. For example, one can value an improvement in a lake's water quality by estimating consequential changes in the market prices of properties around the lake.

**Damage Function Approach** is based on the assumption that the value of a change in an amenity (such as the economic impact of a flood) is approximately equal to the cost of repair damages, loss in profits, and decrease in standards of living associated with the change in the amenity. The approach is thought to provide imprecise benefit estimates because it implicitly assumes that no remedial actions are taken and there is no accounting for market effects. For example, a reservoir can reduce flood damages. The value of the reservoir's flood protection benefit, based on the damage function approach, is the reduction in expenditures made to repair flood damages. But, with reduced risks of flooding, individuals will reduce expenditures on flood-damage prevention. This cost savings is also a flood protection benefit.

**Replacement Cost Method** assumes that the welfare effect of a loss in an amenity is approximately equal to any increase in expenditures that would subsequently restore the amenity. For example, the loss of safe drinking water from one's well is approximately equal to one's subsequent purchases of bottled water. This approach is made difficult because changes in expenditures may not be observed, and remedial action may not be taken.

**Averting (Defensive) Expenditure Method** assumes that the welfare effect of public actions taken to reduce or prevent impacts to an amenity (e.g., reduce downstream flood damages) is approximately equal to the dollar amount individuals pay to reduce or avoid the impacts (such as their cost of sandbagging floodwaters). Some studies argue that this approach overstates benefits (Freeman, 1993; Ribaudo, 1989). Although, in practice, it can be difficult to isolate the portion of expenditures that are attributable to averting damages, the approach continues to be used (Whitehead, 2005).

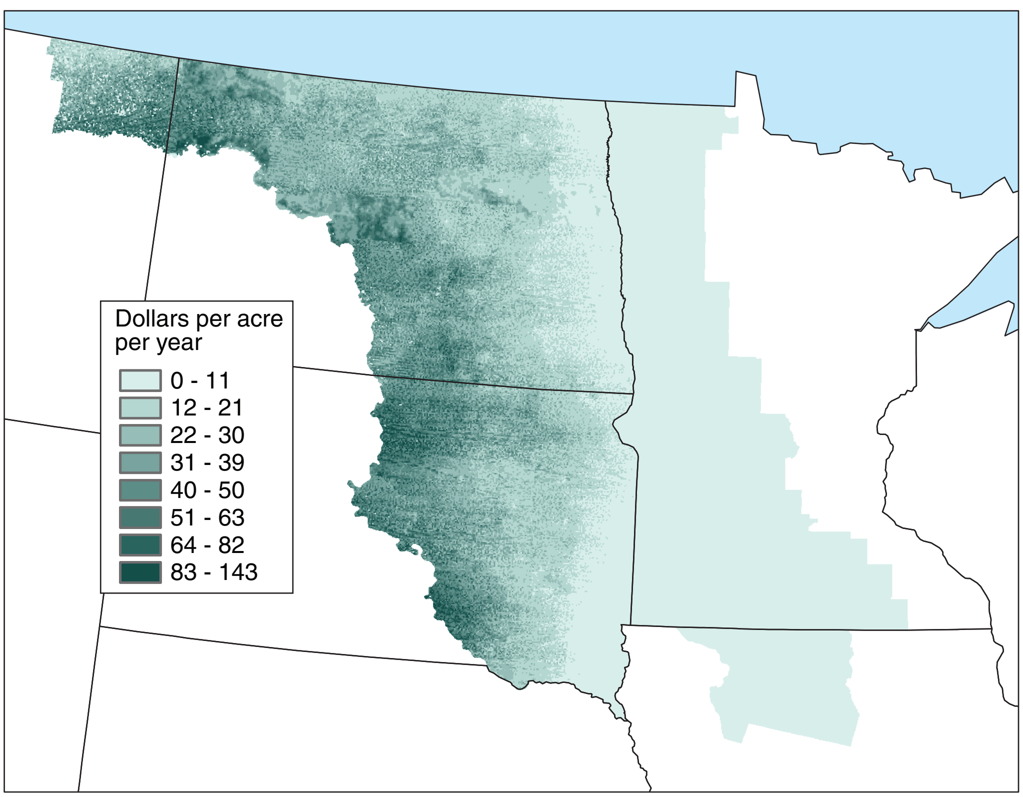
**Benefits Transfer** is a valuation approach used to generate benefit estimates based on the results of past research. There are (roughly) three types of benefit transfer approaches. The value transfer approach reviews the economic literature for per-unit values (or average values) that are then used as a measure of value. The second approach uses past benefit functions and their results to generate more generalized benefit or willingness-to-pay functions. And meta-analyses estimate benefit functions using findings of past research as independent variables. While this approach is convenient, it has imposed substantial errors (Brander et al., 2006).

For this analysis, we developed a biophysical model and applied findings of earlier research to generate spatial estimates of the effects of new wetlands on duck populations (an ecosystem service). Then, using findings of past research and reasonable assumptions, we linked changes in duck populations to changes in hunting quality (an amenity). Finally, we used benefits transfer to tie an economic value to changes in hunting quality. In the PPR, annual per-acre values of the effects of new wetlands on hunting quality ranged from near $0 to $143 per acre, with the highest values in the PPR's western fringe (fig. 3).

### *The biophysical models: Duck populations*

Duck population growth is constrained by the availability of breeding habitat and factors affecting nest success (Hoekman et al., 2002). In other words, an increase in wetlands that support duck reproduction will increase duck populations (a wetland service). We developed models that generate spatial estimates of expected increases in numbers of hatchlings (of the five dominant duck species) produced by new wetlands in the PPR. The models were constructed using a combination of earlier models, findings of past research, and available data. The models simultaneously predict the number of duck nesting pairs that are likely to take residence in a new wetland and the probability that a nest is successful (that is, the eggs hatch), given its location and the surrounding vegetative cover (for details, see appendix 3). To determine the subsequent change in duck populations, we applied findings of Krapu et al. (2000) and Nichols and Hines (1983), which suggest that about 50 percent of all duck hatchlings reach adulthood.

Figure 3
**Value of new wetlands' effects on duck-hunting quality in the Prairie Pothole Region**




Source: USDA, Economic Research Service analysis of wetlands' effects on hunting quality (appendix 3) and a "per-duck" consumer surplus estimate of $106.

The effects of new wetlands on hatchling populations were found to vary substantially across the PPR, with the greatest impacts in the Dakotas along the Missouri River. We found that the effect of a new wetland can range from 0 to over 55 hatchlings (27.5 adult ducks) per wetland acre.

### *Valuation of amenities: Duck hunting benefits*

Multiple studies evaluated the economic value associated with duck hunting. But, only three generated estimates of consumer surplus that result from an improvement in the quality of duck hunting on a scale suitable to analyses of national or regional conservation policies and programs (table 2).[16] Each of these studies defines hunting quality (the amenity) as the number of ducks bagged. Across these studies, willingness-to-pay estimates for a marginal improvement in hunting quality range from $15 to $140 per duck (in 2011 dollars). These studies have limitations (see box "Estimating the Value of One More Duck"). Based on our assessment of this literature, we use a value toward the upper end of this range ($106) and discuss how estimates and their implications are affected by lower duck-value estimates.

### *Linking services to amenities: Hunting quality*

With ducks bagged as a proxy measure of hunting quality, we used prior research and reasonable assumptions to link changes in duck populations to changes in ducks bagged. Based on mallard harvest rates of approximately 10-12 percent and a blue-wing teal harvest rate of approximately 2.5 percent (U.S. Fish and Wildlife Service, 2012), we applied what we believe to be a conservative harvest rate of 5 percent. In other words, each additional duck has a 1-in-20 chance of being bagged. Using an amenity value of $106 per bagged duck, the expected value of an additional duck is approximately $5.00, or $2.50 for each additional hatchling. Multiplying the hatchling value by a wetland's effect on hatchlings, we find that, within the PPR, the annual per-acre values of wetlands duck-hunting benefits range from near $0 to $143, with the highest values in the western fringe of the PPR. Both the value of bagging a duck and the harvest rate directly affect the per-acre value of a wetland. For example, if one were to apply a $26 per-duck value (which is about 25 percent of the value we applied), per-acre values would fall by 75 percent—from near $0 to $46 per acre. If one were to apply a 10-percent harvest rate, per-acre benefits would double. Applying other per-duck values and harvest rates is equally straightforward. In all cases, the pattern of relative values shown in figure 3 remains unchanged.

Table 2

**Studies that estimate the change in consumer surplus associated with bagging an additional duck**

| Study | Valuation method and type of data | Consumer surplus per duck bagged |
|---|---|---|
| Hammack and Brown (1974) | Time series data on hunting activities | $15 to $34 with a mean value of $24.50 |
| Charbonneau and Hay (1978) | Survey of hunting activities of hunters in the Mississippi River flyway | $106 |
| Mackenzie (1993) | Survey of Delaware hunters on hypothetical hunting options | $132 to $140 |

Note: All values are reported in 2011 inflation-adjusted dollars.
Source: USDA, Economic Research Service.

[16]Suitable data for analyses of hunter demands require population surveys with questions that relate to hunting opportunities and details on activities (see box "Estimating the Value of One More Duck").

## Estimating the Value of One More Duck

Our review of the literature revealed five core studies with objectives that include estimating hunters' demand for duck hunting.[1] Three studies specified demand as a function of (among other things) hunting quality (see table 2). These studies derive hunters' willingness to pay for a marginal change in duck hunting quality by differentiating the demand function with respect to hunting quality—bagging one more duck. The first (Hammack and Brown, 1974) used time-series data on hunting behavior in the Pacific Northwest. It estimated the gain in consumer surplus of bagging an additional duck to be between $15.38 and $33.62 (in 2011 inflation-adjusted dollars).

The second (Charbonneau and Hay, 1978) used cross-sectional data from the 1975 National Survey of Hunting, Fishing, and Wildlife Associated Recreation (NSHFWAR) to estimate demand as a function of hunting quality, experience, costs (travel expenses and the opportunity cost of travel time), and income. Based on their estimated demand function, the authors found that hunters are willing to pay, on average, $106 to bag an additional duck on a single day (see table 2).

The third study (Mackenzie, 1993) used data from a survey of Delaware duck hunters where hunters were asked to choose from alternative hypothetical sites to estimate hunters' demand for duck and goose hunting. Similar to the approach of Charbonneau and Hay (1978), demand was estimated as a function of travel cost, travel time, the site characteristics, hunting partners, and site congestion. Mackenzie found that, on average, hunters' willingness to pay for an additional duck bagged ranged from $132 to $140 (see table 2).

Two other studies have directly estimated demands for duck hunting; neither estimated WTP for improved hunting quality. Each generated per-day consumer surplus estimates, as did Charbonneau and Hay (1978). We therefore compare the day-value estimates to gain some insight into the consistencies of the estimated demand functions. Charbonneau and Hay (1978) estimated per-day consumer surplus to be $121. Brown and Hay (1987)—employing the 1980 NSHFWAR data—estimated per-day consumer surplus to range from $64 to $82, which is about 65 to 80 percent of the Charbonneau and Hay (1978) estimate. However, Brown et al. (1978)—using the same data as Charbonneau and Hay—estimated average per-day consumer surplus to be $122, which supports the findings of Charbonneau and Hay (1978).

We chose not to use Hammack and Brown (1974) because, unlike subsequent research, they failed to account for the opportunity cost of time—the opportunity cost continues to be recognized as a critical part of travel cost (Larson and Lew, 2014; Menon et al., 2014). Additionally, Gascoigne et al. (2011) rejected the Hammack and Brown (1974) model because, upon the use of more recent values of the independent variables, they felt the model generated values that were unacceptably low (less than $4 per duck).[2]

We applied the Charbonneau and Hay estimate of $106. The more recent Mackenzie (1993) results suggest this value might be low. Conversely, our day-value comparison with the Brown and Hay (1987) estimates suggests the Charbonneau and Hay (1978) estimate might be high.

The age of these studies raises questions: Has the public's willingness to pay changed over time? Has the increase in the U.S. population increased the number of duck hunters and the number of people benefiting from improved hunting quality? Have behavioral models or empirical methods improved in such a way that duck hunting demand might be better estimated? These same questions arise in most meta analyses. As with authors of previous studies, we assume that, after adjusting for price-level changes, the reported values are reasonable. But given our uncertainty of this estimate, we also use other duck values to illustrate the sensitivity of our results.

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[1]These five studies estimate hunters' demand for hunting across wide regions—they estimate the aggregate demand within their study area. Many other studies estimate hunters' demand at a single site or across substitute sites, such as game refuges in California. Demand at each site depends on the size and interest of the surrounding population, travel costs, the quality of hunting at surrounding sites, and other location-specific factors. If the results of a substantial number of site-demand studies scattered throughout the relevant region were available, it may be possible to estimate an aggregate demand based on the results of the site-level studies. However, instead of the studies being scattered throughout, the majority of the studies focus on sites on public lands in areas where hunting is popular.

[2]Gascoigne et al. (2011) subsequently moved from a marginal benefit analysis to an aggregate day-value approach.

### Nitrogen removal: What the study found

Wetlands can improve water quality by removing chemicals before they reach streams, rivers, and other waterways. Of particular interest is the ability of wetlands to filter out reactive nitrogen (N).[17] Reducing nitrogen loadings in water provides various amenities:

- Improvements in the quality of municipal intake waters
- Cleaner water for swimming, boating, and recreational fishing
- Increased profitability of commercial fisheries

Confined by the limitations of data and models, we evaluated N removal by new wetlands in the Upper Mississippi and Ohio River watersheds—this area is the primary source of N affecting the hypoxic zone of the Gulf of Mexico, where excess nutrients have depleted oxygen needed to support marine life. Additionally, this region has opportunities for wetland restoration because 80 to 90 percent of prior-existing wetlands were drained. But we could not link the change in nitrogen loadings to an impact on an environmental amenity, nor could we find or estimate a monetary value for impacts on amenities. We therefore focused on N removal and found that rates varied substantially, ranging from 11 pounds to nearly 1 ton per wetland acre per year (see fig. 4).

### The biophysical model: Nitrogen removal

Building on available models and data, we generated estimates of the expected quantity of N removed by a restored 10-acre wetland. These models work with hydrology, inflow water quality, and weather and climatic conditions at grid points throughout much of the Upper Mississippi and Ohio River watersheds.[18] To capture the effects of daily and annual variations in precipitation levels and intensity on nitrogen movement and wetland effectiveness, we ran 20-year simulations (see appendix 4).

### Economic models of amenities affected by nitrogen inflows

We found two large-scale economic analyses of the benefits of reducing N loadings; both relied on reduced-form models. First, in an attempt to value changes in Gulf of Mexico hypoxia, Diaz and Solow (1999) used 30-year time-series and geographical data to estimate economic impacts on commercial fisheries and recreational fishing activities as a function of the size of the hypoxic zone and other factors and found no statistically significant results.

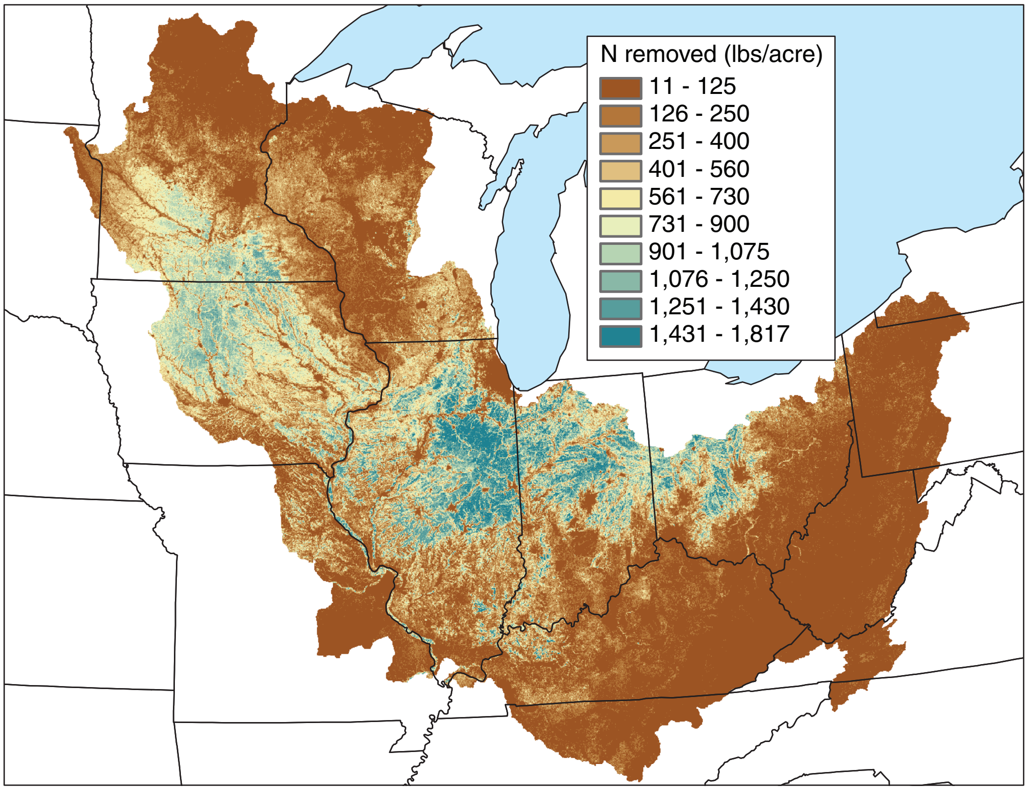
Second, in an analysis of municipal water treatment costs, Ribaudo et al. (2011) found that nitrogen has a statistically significant impact on costs. They reported municipalities' longrun nitrogen removal costs ranging from $20 to $36 per million gallons of treated water in those areas where treatment plants are necessary. Note that this cost estimate is a function of the volume of water and

[17]Reactive nitrogen is a general term used to represent the set of nitrogen compounds that support plant growth. Other chemicals include phosphorous, herbicides, and insecticides. Due to a paucity of data on the prevalence and impacts of these chemicals, these issues are left for future research.

[18]Other wetlands are known to remove nitrogen from surface waters, but the complexities of very similar wetlands' hydrology make it difficult to predict effects of similar wetlands. For example, in an analysis of depressional wetlands near the Chesapeake Bay, Denver et al. (2014) found that the depth of the aquifer below the wetland is the largest determinant of nitrogen removal. Information on aquifer depth is not readily available, making it difficult to predict the nitrogen removal rate of a restored wetland in some areas.

Figure 4

**Amount of nitrogen removed annually by a new wetland in the Upper Mississippi and Ohio River watersheds**




Source: USDA, Economic Research Service.

not the quantity of nitrogen. Even so, the study's findings suggest that a change in nitrogen inflow to streams affects water treatment costs.

At the national level, the U.S. Environmental Protection Agency Science Advisory Board (2007) examined both the benefits and co-benefits of nitrogen conservation practices, including those at wetlands. The Board found evidence of economic benefits of N reductions but concluded that co-benefits (such as the values of consequential decreases in soil erosion) may exceed the nitrogen reduction benefits.

### *Linking services to amenities: The value of reduced N inflows*

Overall, we were unable to link spatial changes in N flows into streams to the subsequent outflows to the Gulf of Mexico. But Rabotyagov et al. (2014) found a statistically significant link between N inflow to the Gulf of Mexico and the size of the hypoxic zone, using a simple statistical model and time-series data.

Overall, evidence of the effects of the Gulf of Mexico's hypoxic zone on ecosystems is mixed. Bianchi et al. (2010) found no evidence that discharge from the Mississippi River adversely impacted Gulf of Mexico fisheries. Consistent with the findings of Diaz and Solow (1999), other studies found no clear link between hypoxia and changes in fishery landings in the Gulf (Rabalais and Turner, 2001). In fact, Bianchi et al. (2010) note that there appears to be some effect on the distributions of mobile fish species and phyto- and zooplankton that prosper in the nutrient-rich waters that add to, what appears to be, an additional food web.

Still, there are grounds for concern about hypoxia in the Gulf of Mexico. For example, the Federal-State Mississippi River Gulf of Mexico Watershed Nutrient Task Force (2013) noted that "low dissolved oxygen in the Gulf is a serious environmental concern that can impact valuable fisheries and disrupt sensitive ecosystems."

The findings of analyses of waters in the Gulf of Mexico must not be viewed as representative of other hypoxic zones. The hypoxic zone of the Gulf of Mexico lies within a great expanse of deep waters while, in most other cases, hypoxic zones are in shallow waters in bays and along shorelines where ecosystems are easily impaired and damaged (Bianchi et al., 2010).

## Species protection: What the study found

The public's willingness to pay to protect species reflects the following:

- A desire for opportunities to view the species
- The ethics of protecting resources for future generations
- Uncertainty that there may be subsequent domino effects on other species if one or more species or their habitats are lost

This analysis considers only wetland-associated species identified by NatureServe as imperiled in 2014.[19] In 2014, approximately 597 wetland-associated species were listed as imperiled (217 are listed as critically imperiled) across the entire range for each in the United States.[20] The number of wetland-associated imperiled species within a county ranges from 0 to 23 (fig. 5a).

### *The need for a biophysical model: Species protection*

An appropriate biophysical model would quantitatively link the addition of a wetland to the affected species. We obtained data from NatureServe (2014) that lists each wetland-associated imperiled species and the counties where each exists. The data are insightful but do not indicate whether a species' survival is affected by an additional wetland—other habitat conditions, such as water quality, predation, and migratory habitat, may have an effect. Even in counties with no wetland-associated imperiled species, it is possible that an additional wetland could have a positive impact on imperiled species—that is, expanding wetland habitat could help imperiled species in neighboring or downstream counties.

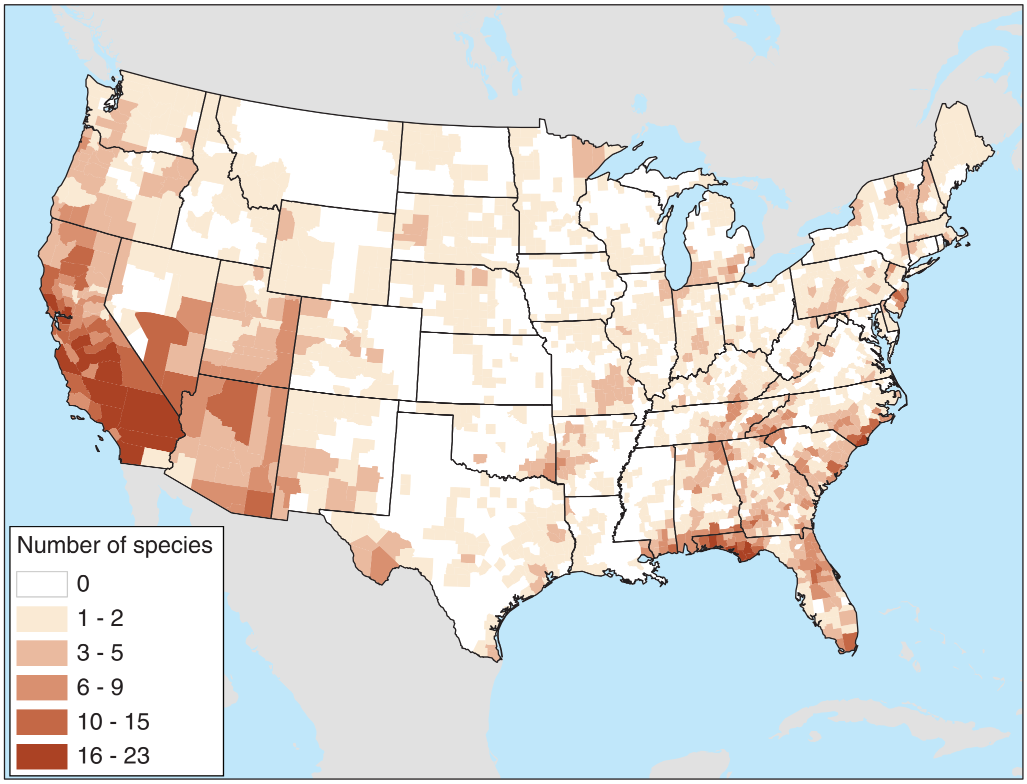
### *Valuation of amenities: Species protection*

Some view species protection as a moral or ethical issue and object to placing a value on protecting a species. However, Shogren et al. (1998) point out that, while most Americans believe in fully

[19]The NatureServe data used in our analysis include all species that have a NatureServe Global Conservation Status Rank of G1or G2. A rank of G1 indicates that a species is critically imperiled across its entire range. A rank of G2 indicates the species is imperiled over its range. Rankings are based on various biological and external factors that may affect the long-term viability of a species (Regan et al., 2004). For details on species classifications, see appendix A, page 46 in www.natureserve.org/sites/default/files/publications/files/natureserveconservationstatusfactors_apr12_1.pdf

[20]ERS analysis of NatureServe 2014 data (NatureServe, 2014).

Figure 5a
**Wetland-associated imperiled species, by county**




Source: USDA, Economic Research Service using data from NatureServe, 2014.

protecting endangered species, they are unwilling to bear the cost of increasing the strength of the 1973 Endangered Species Act. Because public funding for species protection is limited, decision-makers who prioritize wetland conservation efforts consider tradeoffs in species' protection so that, implicitly or explicitly, species are being valued (Roughgarden, 1995).

Past research suggests that the public tends to have a higher willingness to pay to protect so-called "charismatic" species, including many large birds and mammals, than to protect others (Loomis and White, 1996; Brander et al., 2006; Richardson and Loomis, 2009; and Woodward and Wui, 2001). In developing data for their meta analysis, Richardson and Loomis (2009) collected published species valuation studies. Most of the studies they found used contingent valuation, where respondents are asked a series of questions that are designed to reveal their willingness to pay to protect one or more species. However, the reported values are often based on a generalized concept of protection that cannot be directly used to calculate values of marginal improvements in species' survival.

### *Amenities provided: Wetland effects on imperiled species*

An increase in a species' survival probability is a quantitative measure of an increase in an environmental amenity (see 5-8 in U.S. Environmental Protection Agency (2014)). Research has yet to develop biophysical models that quantify new wetlands' effects on species survival probabilities (e.g., the amenities). Without a measure of new wetlands effects, estimates of the values of changes in species' survival probabilities offer little insight into the relative importance of different wetlands' effect on imperiled species.

In the absence of biophysical models, we have generated proxies to provide insight into the effects of wetland restoration on imperiled species. The proxy is the number of wetland-associated imperiled species in a county. This approach has been used in past studies—counties with high imperiled or endangered species counts are considered "hot spots" (Dobson et al., 1997; Ando et al., 1998). Szentandrasi et al. (1995) used this approach to assess how one might target CRP enrollments.

An analysis of NatureServe data shows that imperiled species were located in 1,745 counties (about 57 percent of all contiguous U.S. counties) in 2014, and that 1,366 counties have no wetland-associated imperiled species. Concentrations of imperiled species are greatest in the Southwest (as many as 23 per county) and are spread throughout much of the Southeast (fig. 5a).

Restricting the analysis to vertebrate species (such as birds, amphibians, and mammals), which tend to receive the most public attention, we found a total of 76 imperiled wetland-associated species (28 of which are critically imperiled) and numbers per county ranging from 0 (in 2,010 counties) to 8. Southern California and neighboring lands and Appalachia are the hot spots (fig. 5b).[21]

These findings must be cautiously interpreted because there are no measures of the effects of a new wetland or the values of effects on different species. Findings do, however, provide some perspective on where new wetlands might benefit a greater number of imperiled species.

## Flood protection: What the study found

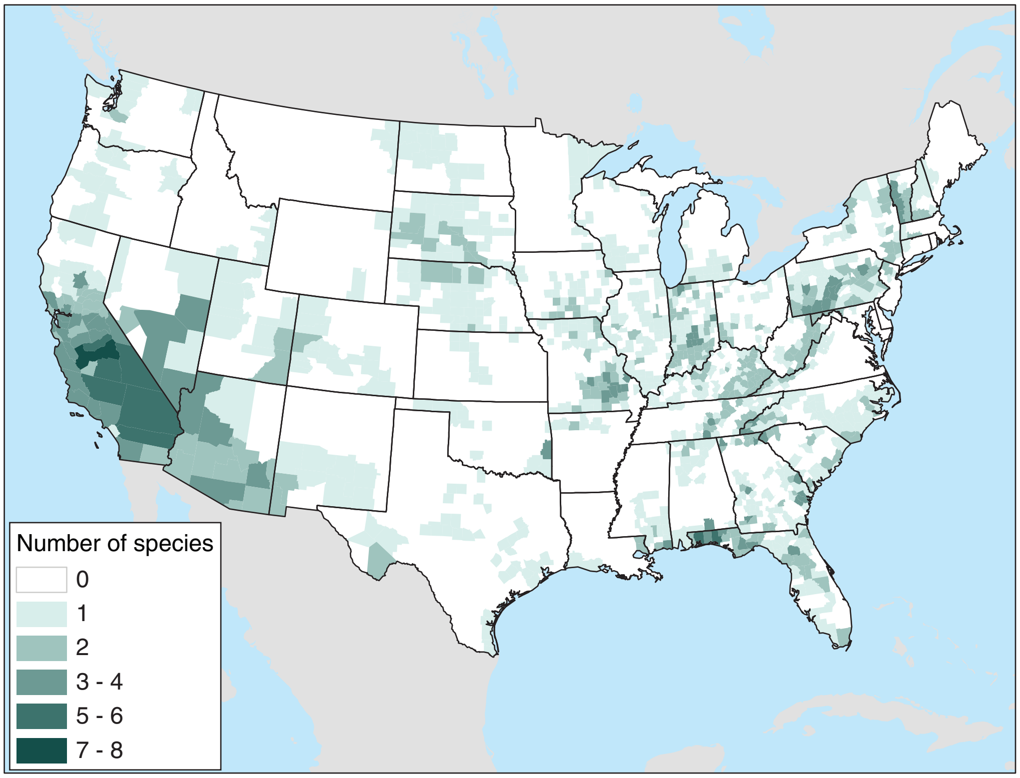
Society is willing to pay to reduce flooding because of private and public benefits including damages:

- To reduce personal risks of damage to property and health and a disruption of daily lives
- To reduce public expenditures on disaster relief
- To avoid damage to public infrastructure
- To maintain healthy communities
- To adhere to ethics of protecting the health and property of others

Wetlands provide protection by storing water, slowing its movement, and reducing storm surges associated with heavy precipitation and rapid snow melts. Losses caused by floods include both human life and property. Floods damage private structures and public infrastructures, destroy crops, and wash away farmland. According to the U.S. Department of Commerce, National Oceanic and Atmospheric Administration (NOAA), annual property damages attributed to flooding usually lie between $1 billion and $10 billion, but a spike of $50 billion occurred in 2005 due to a large number of storms and the especially severe damage wrought by Hurricane Katrina.

[21]Counties where imperiled wetland-associated species are prevalent may not be important to agricultural policy if there are few opportunities to restore wetlands on agricultural land.

Figure 5b
**Wetland-associated imperiled vertebrae species,* by county**




*Includes birds, mammals, amphibians, and reptiles.
Source: USDA, Economic Research Service using data from NatureServe, 2014.

## *The biophysical model: Floodwater storage*

In a comprehensive analysis, Bullock and Acreman (2003) reviewed 169 studies of wetlands' hydrologic performances, including their effects on flooding. Findings of these studies varied widely—in some cases, wetlands increased flooding. They conclude that, while wetlands play a role in flood protection, comparisons of wetlands' effectiveness must be given careful consideration.

A wetland's "bounce"—its ability to store additional water as rivers reach peak flows—can help determine whether the wetland can control flooding. A wetland's bounce is usually small (Bullock and Acreman, 2003). However, bounce can be increased if water level can be controlled. For example, in a site-level analysis, Bengtson and Padmanabhan (1999) used a hydrologic model to estimate the relationship between reduced peak flow and storage bounce in the Red River Watershed of North Dakota. They found that, unless modified, wetlands in the watershed did little to reduce peak flows because, as river levels rose, wetlands filled. But, if wetlands were allowed to drain until rivers began to peak, a 1-foot impediment placed on all wetlands (creating a 1-foot storage bounce across the wetland acres) could reduce damages from high-frequency flood levels by 3.8 percent; a 2-foot impediment could reduce damages by over 5 percent.[22]

[22]Wetlands can reduce flood levels from short, heavy downpours when wetland water levels are relatively low, and they can capture a portion of the surge in water flows.

*Valuation of amenities: Expected reductions in flood damages*

Five studies in the literature estimated the economic value of wetland flood protection benefits. The studies do not offer values directly transferable to wetlands restored on agricultural lands for one or more of the following reasons: they focus on wetlands in close urban proximity with limited agricultural lands; they focus on coastal areas where agricultural acreage is limited; they evaluate wetlands with flow impediments; and they aggregate wetlands with other land uses. However, the studies are likely to illustrate upper-bound ranges of flood protection values.

Gupta and Foster (1975) estimated the flood control benefits of wetlands in the Charles River Basin of metropolitan Boston—a major urban area. Using Army Corps of Engineers data, they found expected avoided damages to be $56 to $440 per wetland acre (inflation adjusted to 2011 dollars) per year. Values such as these may be applicable to wetlands on agricultural lands but in limited areas with considerable housing or urban development.

NOAA provides statistics on some U.S. coastal wetlands' services, including estimates of coastal wetlands' flood protection benefits that were originally generated by Costanza et al. (2008) (see http://stateofthecoast.noaa.gov/wetlands/welcome.html). Costanza et al. reported benefits of near $0 to $29,600 per acre per year, with some of the highest benefits in Louisiana. Conversely, Farber (1987) estimated benefits of flood protection by coastal wetlands in Louisiana to be substantially less (i.e., less than $50 per acre). Again, values such as these may be applicable to wetlands on agricultural lands in coastal areas.

Shultz and Leitch (2003) estimated the flood protection benefits of restoring 2,700 acres of wetlands—with flow impediments—in the Maple and Red River Watersheds of North Dakota. They found that when wetland outlets were controlled (e.g., water was captured only when flows were peaking, consistent with Bengtson and Padmanabhan (1999)), expected annual benefits ranged from $12 to $25 per acre.

The U.S. Army Corps of Engineers (2013) estimated flood damage reduction benefits to downstream urban areas due to the CRP's wetlands and riparian buffers. This pilot study focused on a small Iowa watershed (Indian Creek). It used several hydrologic models, simulated weather events, and a variety of CRP coverage scenarios (where each scenario differed in the quantity of CRP acreage and the distributions between wetlands and riparian buffers). The study found that, when rainfall events were short but very heavy, acres under CRP wetland practices can reduce property damage by 11 percent (although acres in riparian buffers are more cost effective). When the methods of this study were applied to a larger Iowa watershed (Cedar River), there was no measureable relationship between the CRP's wetland and riparian buffer acreages and the change in flood stage.

*Amenities provided: Reduced flooding due to wetlands*

The dollar-per-acre flood protection benefits reported in Shultz and Leitch (2003) and other studies embody the link between the effects of wetlands on flooding and flood damages—they are reduced-form analyses in which the benefit estimates are dependent on physical and economic parameters. We found too few studies in the literature that generated spatial estimates of wetlands' flood protection benefits within any region to enable us to develop benefit models through benefits-transfer analyses.

## Net greenhouse gas emissions: What the study found

Many countries are seeking to reduce the growth in atmospheric concentration of greenhouse gases (GHG), primarily by reducing emissions. Some countries are also engaged in efforts to remove GHGs from the atmosphere, commonly through carbon sequestration.

Wetlands are known to reduce atmospheric concentrations of carbon dioxide through carbon sequestration. But wetlands are also known to emit GHGs, primarily methane ($CH_4$) and nitrous oxide ($N_2O$). Both of these are considered "hot" and can more than offset the effect of $CO_2$ removal.

Past research that has reviewed and evaluated findings of studies worldwide (Kayranli et al., 2010), within North America (Bridgham et al., 2006), and within the tropics and subtropics (Mitsch et al., 2012) concluded that the effects of wetlands on GHG emissions are questionable, as carbon sequestration may be more than offset by methane and nitrous oxide emissions. These studies illustrate the importance of accounting for net GHG effects, but they do not provide measures of the GHG effects of different types and ages of wetlands. Similarly, many studies have estimated carbon sequestration and carbon storage by different types of wetlands (for an overview, see Bernal and Mitsch, 2012), but most do not measure net GHG impacts.

Euliss et al. (2006) and U.S. Fish and Wildlife Service (2005) evaluated the net GHG effects of wetlands in the PPR and Mississippi Alluvial Valley (MAV) (see fig. 1), and findings from these studies are used in this analysis along with estimates of the marginal social cost of carbon developed by a Presidential Commission (U.S. Government, 2010). Based on these data, the annualized carbon sequestration benefit of a new, permanently restored wetland is $55 per acre in the PPR and $129 for an MAV wetland (table 3).

### *Net greenhouse gas emissions: The biophysical models—GHG impacts*

Past research provides estimates of the net GHG impacts of wetlands in two regions: the PPR and MAV. Euliss et al. (2006) estimated that, 5 years after a wetland in the PPR is restored, 7.8 metric tons per acre of carbon is held in the living biota above and below ground—an average sequestration rate of 1.56 metric tons per acre per year. In subsequent years, carbon sequestration was estimated to average 0.34 metric tons per acre per year. Additionally, while wetlands of the PPR were estimated

Table 3
**Annual average per-acre carbon sequestration benefits of Prairie Pothole Region and Mississippi Alluvial Valley wetlands**

| Wetland regions* | SRD=3 r=4 | SRD=5 r=4 | SRD=2.5 r=4 | SRD=3 r=5 | SRD=3 r=3 |
|---|---|---|---|---|---|
| | | | *$/acre** | | |
| PPR | 1,378 | 371 | 2,160 | 1,251 | 1,536 |
| MAV | 3,222 | 930 | 4,955 | 2,776 | 3,667 |
| Annualized values | ---------------------- | 4% | ---------------------- | 5% | 3% |
| PPR | 55 | 15 | 86 | 62 | 46 |
| MAV | 129 | 37 | 198 | 139 | 110 |

PPR=prairie pothole region; MAV=Mississippi Alluvial Valley: SRD=social rate of discount (%); r=market discount rate (%).
*2011 dollars.
Source: USDA, Economic Research Service.

to release methane and nitrous oxide, the release rates were estimated to equal the releases on the lands' prior wetland restoration, meaning that the restored wetlands led to no change in the releases of these gases (Gleason et al., 2009; Gleason et al., 2005; Eagle et al., 2012).

In a second study in the PPR, Gleason et al. (2008) estimated carbon in above- and below-ground vegetation after 10 years to be about 11.13 metric tons per acre, which is about 1.6 metric tons per acre greater than the amount estimated by Euliss et al. (2006). We applied the Euliss et al. estimates because they are more conservative and they account for the effect of time on sequestration rates.

Within the MAV, carbon accumulates primarily in the vegetative biomass (Faulkner et al., 2008). In 2005, the U.S. Fish and Wildlife Service (FWS) reported that vegetation on newly restored MAV wetlands held 99 metric tons per acre of carbon after 70 years for an average annual rate of 1.41 metric tons per acre per year (USFWS, 2005). A second FWS analysis, also released in 2005, reports a 70-year average carbon sequestration rate of 1.77 metric tons per acre per year. We applied the more conservative 1.41 metric tons per acre per year. Research on $N_2O$ emissions by restored forested wetlands found no difference from the level released on agricultural cropland (in the first 70 years), while net effects on $CH_4$ releases are unknown but are not expected to have a large effect (Faulkner et al., 2008).

The GHG impacts reported here are only relevant in cases where cropland is converted to wetlands. As with all evaluations of the GHG impacts of conservation practices, the prior use of the land and its effect on carbon loadings and releases of GHGs must be considered as it defines baseline conditions (see box "The Value of GHG Impacts Depends on GHG Emissions Over Time"). For example, pasture and hay land usually hold a greater stock of carbon in above- and below-ground vegetation than cropland. Therefore, the GHG impacts of wetlands restored on pasture or hay land will likely be lower than those of wetlands restored on cropland.

### *Valuation of amenities: Net greenhouse gas emissions*

To meet Executive Order 12866, an interagency research team was assembled to generate estimates of the social cost of carbon (SCC).[23] These estimates are measures of the marginal cost imposed by a 1-ton change in atmospheric carbon dioxide in an observation year. Because the marginal economic effects increase as GHGs accumulate, the SCC increases over time (see table A1 in U.S. Government, 2010).[24] The Executive Order requires agencies to use the estimates in cost-benefit analyses of policies and actions that have marginal impacts on GHGs (for details, see U.S. Government, 2010).

The SCC estimates—calculated using the damage function approach—are based on agreed-upon values related to many controversial issues, including the value of a human life, the expected rate of GHG accumulation, technological advances that offset impacts, and—perhaps the most economically controversial—the rate at which future damages (or adverse impacts to individuals' life and

[23]Executive Order 12866 required agencies "to assess both the costs and the benefits of the intended regulation and, recognizing that some costs and benefits are difficult to quantify, propose or adopt a regulation only upon a reasoned determination that the benefits of the intended regulation justify its costs" (U.S. Government, 2010).

[24]These social cost of carbon (SCC) estimates are somewhat higher but not out of line with those reported elsewhere. For example, Nordhaus (2014) estimated the SCC to be $18.60 per ton in 2015 while the International Panel on Climate Change (IPCC) reported an SCC of $23.80. But Nordhaus (2014) found the SCC increase to be faster (3 percent annually), reaching $52.30 in 2050, compared with the IPCC estimate of $44.90.

## The Value of GHG Impacts Depends on GHG Emissions Over Time

The carbon cycle associated with an agricultural practice is the annual change in atmospheric greenhouse gases (GHGs) over time, given what they would have been. This means that one must consider lands' GHGs effects with and without the practice, even after the practice is terminated.

Each social cost of carbon (SCC) estimate applied here is defined as the present-value sum of future damages of the climate change impact due to a marginal change in atmospheric GHG concentrations (see U.S. Government, 2010). The SCC estimates increase over time because, as GHG concentrations increase, marginal damages increase.

Because the SCC rises over time (though the present value of future SCC estimates fall), it is critical to account for changes in GHG loadings on a year-by-year basis. This means one must account for effects after a conservation project's life. By doing so, one finds that permanent conservation practices can generate far more benefits. For example, if a wetland restored (or tillage practice begun) at the beginning of 2013 sequesters 1 ton of carbon per year, its GHG benefits (assuming no impacts on other GHGs) would be $24.71 in 2013, $25.28 in 2014, etc.[1] If the wetland is returned to agricultural production in 2015 and releases 1 ton of carbon each year, the damage due to the ton released in 2015 would be $25.82 and $26.00 for the ton released in 2016. The net GHG benefit of the wetland would be $2.09, when discounted to 2013 ($2.09=24.74 +25.28*(1.04)-1 – 25.82*(1.04)-2 – 26.37*(1.04)-3). Should the same conditions hold and the wetland have a 10-year life, the net present value of the GHG impacts would be $34.00. Finally, if the wetland was left undisturbed, the net present value of the GHG impact would be $623—nearly 20 times the value of the 10-year life and 300 times the value of the 2-year wetland.

[1]Values are from U.S. Government (2010) in 2011 dollars.

standard of living) are discounted, or social rate of discount (SRD) (U.S. Government, 2010). Some observers believe that the SRD should be 0 so that, for example, the value of a lost life in the future is the same as a lost life today; others believe a market discount rate is appropriate. This study applied SCC estimates that were derived from the agreed-upon SRD of 3 percent. For more on the conceptual differences between market and social discount rates and the sensitivity of the SCC estimates to the SRD, see U.S. Government (2010).

### *Amenities provided: Wetland impacts on GHG*

Each year, the values of a new wetland's GHG-reduction benefits differ due to variations in GHG impacts and the SCC. Instead of listing annual values, this study capitalized them using a 4-percent market rate of discount.[25] We found that the present value of the GHG impacts of a restored wetland in the PPR is $1,378 per acre and $3,222 for an MAV wetland (assuming the wetland was restored in 2013). Our estimates assume there will be no changes in wetlands' GHG benefits after 2050 because there are no SCC estimates beyond this date. Had there been estimates for more years, our estimates of the GHG benefit of the restored and then undisturbed wetlands of the PPR and MAV would probably have been greater. To compare GHG benefits with our estimates of annual benefits and costs, we annu-

[25]The market discount rate is appropriate because we are evaluating investments in wetland resources, not the social welfare effects of losses in lives and human welfare. We apply a 4-percent market rate of discount in this analysis, the longrun return on AAA bonds (U.S. Federal Reserve, 2012).

alized total benefits across the wetlands' permanent lives using a 4-percent discount rate (as used in our cost analysis) and found regional values of $55 and $129 per acre, respectively (see table 3).

These benefit estimates are based on *permanent* restoration. If a wetland is moved back into agricultural production, carbon is released and GHG impacts become negative. For example, the present value of the carbon sequestration benefits of a permanently restored PPR wetland can be 20 times the value of a wetland that is converted to agriculture after 10 years (see box "The Value of GHG Impacts Depends on GHG Emissions Over Time"). All economic analyses of GHG benefits of conservation practices need to account for longevity. Conversely, losses of wetlands eliminate the flow of annual benefits provided, such as improved water quality and better hunting quality, but there are no subsequent damages (e.g., no negative costs imposed).

## Groundwater recharge, sediment removal, and open space: What the study found

No regional studies or data measure or value the effects of wetland restoration on groundwater recharge, sediment removal, or open space. Based on a number of small-scale studies, we concluded that (1) wetlands will sometimes provide positive recharge, sediment removal, and open-space benefits, but the sediment-trapping and open-space benefits can be negative; and (2) the benefits for each must be considered on a case-by-case basis.

### *Groundwater recharge: The analyses*

The available literature indicates that, in some locations, wetlands may increase amenities through groundwater recharge (that is, the flow of water to groundwater supplies). But currently, little information is available on the effect of wetlands on groundwater.

In most cases, wetlands do not recharge groundwater supplies. Some wetlands are wet because they are fed by groundwater, and some have a mutual exchange of water such that wetlands hold water when water tables are high and release water as the water table drops. In other cases, the soils under the wetlands are impermeable (Smith et al., 2011) (for more details on wetland types, see www.water.ncsu.edu/watershedss/info/wetlands/function.html). However, wetlands in floodplains are often permeable and provide recharge that can help maintain flows in dry conditions (Bullock and Acreman, 2003).

The value of water added to groundwater supplies depends on how the water might serve the public. Should the groundwater be a source of drinking or irrigation water, the affected individuals will value reduced pumping costs and any lessening of water-supply constraints. People will, indirectly, value groundwater that feeds lakes and streams if it subsequently increases amenities, such as fishing quality, species protection, and scenic beauty.

### *Sediment removal: The analyses*

Based on the available literature, we were unable to conclude whether sediment trapping by any type of wetland will have net positive benefits. By trapping sediment, wetlands can improve water quality, but the sediment can damage the wetland ecosystems and, over time, fill some types of wetlands and eliminate subsequent wetland benefits (Gleason and Euliss, 1998). Other types of wetlands, such as riparian wetlands, may periodically receive high water flows that flush sediment into deeper waters and waterways; the flushed sediment can have adverse environmental impacts.

### *Open space: The analyses*

We were unable to find studies that report the value of open space in rural settings. This is not unexpected because in rural areas, the public benefits of an additional acre of generic "open space" are likely to be small.[26] However, wetlands—as natural open spaces—may have open space values greater than lands without wetlands. But a few factors may reduce the size of, or offset, this extra value. First, since these wetlands are on private land, they may not be accessible (or visible) to the public, which may reduce or eliminate open-space values. Second, "slippage" may occur. Specifically, a new wetland easement will preserve the open space of land in the easement, but it will not prevent the loss of open space anywhere else. The value of the open space that is preserved by the easement can be offset when land is subsequently developed elsewhere (Lichtenberg, 2011; Irwin and Bockstael, 2004).

[26]Given that rural areas, almost by definition, have low population densities, open space is abundant.

# Targeting Wetland Conservation Efforts

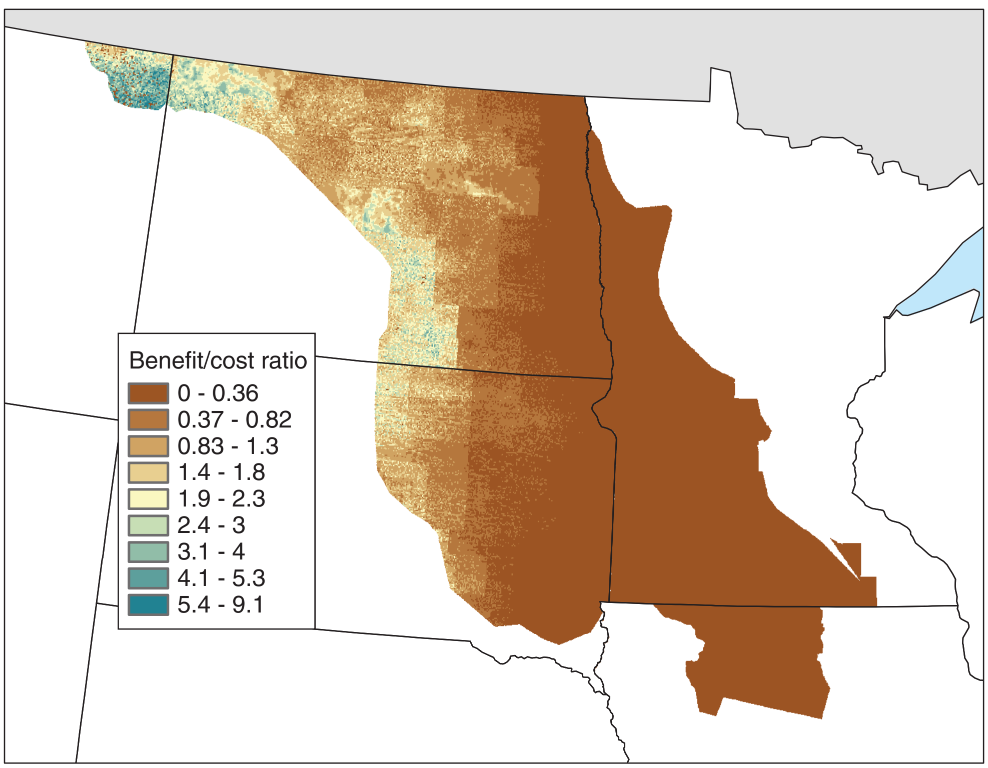
In this section, we overlay the spatial estimates of annual wetland benefits and costs and highlight implications for targeting wetland conservation funding to maximize these benefits relative to costs.[27]

## Duck hunting benefits and costs of prairie pothole wetlands

Duck-hunting benefit-cost ratios are greater than one in about one-third of the PPR and are greatest in Montana and parts of western North Dakota, driven by both lower land costs and higher ecosystem productivity (fig. 6). This finding suggests that, in general, wetland restoration funds spent in the western PPR are likely to be a good investment in that they are likely to provide a positive social rate of return. Note that we have not conducted a site-by-site analysis, so some specific sites might have benefit-cost ratios that may or may not exceed our area estimate.

Figure 6

**Duck-hunting benefits relative to wetland cost in the Prairie Pothole Region**




Note: Based on a benefit estimate of $106 per duck.
Source: USDA, Economic Research Service analysis of the value of wetlands' duck hunting benefits in figure 3 and wetland costs in figure 1.

[27] We were unable to include all costs, or all benefits, in our analysis so the estimated benefit-cost ratios may be too high or too low. However, for those benefits that were estimated, we expect that the spatial variations in benefit-cost ratios are realistic and informative for allocation decisions.

Factors to consider when applying the duck-hunting benefit-cost estimates:

- If the value of new hatchlings differs across flyways, our benefit estimates may be missing some variation in wetland values.

- Other values of changes in duck populations, such as the value to birdwatchers, were not included because we found no measures.

- Wetlands of the PPR provide important duck nesting habitat (50 to 80 percent of all North American ducks nest in the PPR (Batt et al., 1989)). Wetlands in other parts of the country, suitable to nesting, will also increase duck populations and improve hunting quality.

- Because duck populations are constrained by wetlands that can support nesting populations (Reynolds et al., 2007), improving or expanding wetlands in areas where ducks do not nest is not expected to increase duck populations.

- We use a value of $106 per duck. Changing this value (for example, assuming the marginal value of a duck is $26) will cut benefits by 75 percent and reduce the size but not the spatial pattern of the distribution of the benefit-cost ratios. Conversely, if bag rates are greater than 4 percent (a value we feel is conservative), per-acre benefits would be greater. For example, increasing the bag rates to 10 percent would double the benefits amount.

## Nitrogen removal: The cost effectiveness of wetlands of the Upper Mississippi and Ohio River watersheds

The amount of nitrogen removed per dollar spent to restore and preserve a new wetland ranges from 0.15 to 34 pounds (fig. 7). These estimates indicate that nitrogen removal costs, via new wetlands, range from $0.03 to $7.00 per pound. Within the area of study (the Upper Mississippi/Ohio River watershed), wetland restoration in east-central Illinois and southwestern Ohio tends to be most cost effective. If nitrogen reduction was the only environmental goal, these areas would be a good place to target wetland restoration funding.

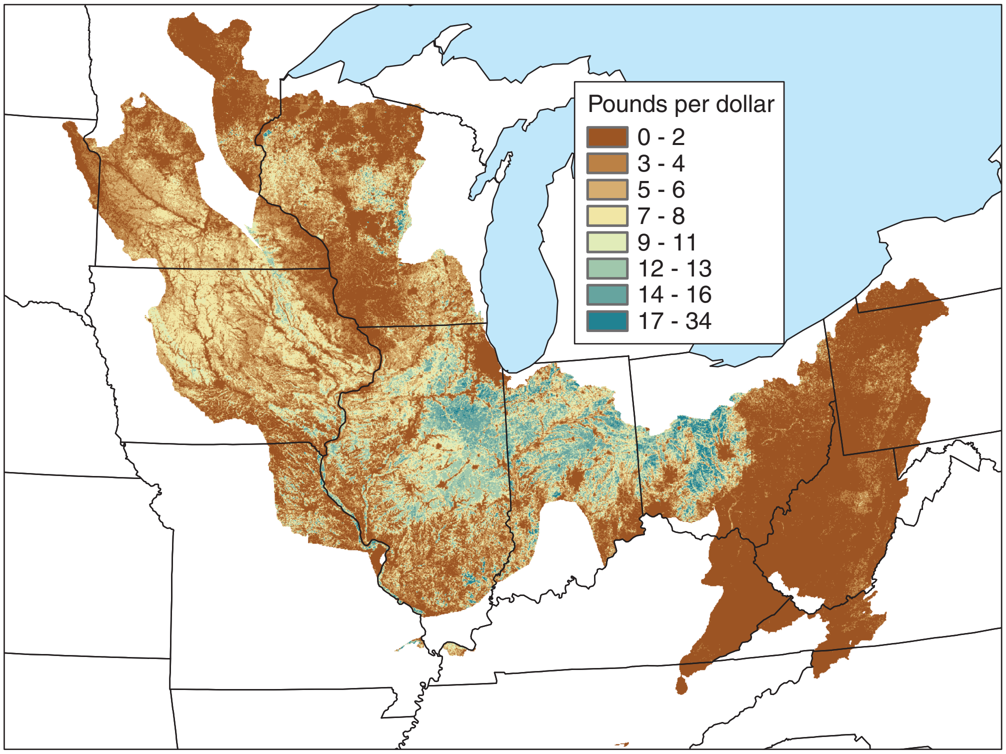
Variations in costs and nitrogen removal rates both are strong drivers of cost-effectiveness. For example, east-central Illinois, southwestern Ohio, and north-central Iowa have some of the highest wetland nitrogen removal rates in the region (see fig. 4). However, high wetland costs in north-central Iowa make wetland restoration in this area less cost effective than in the other areas mentioned (see fig. 7).

Although nitrogen removal is not the only way that wetlands improve water quality, factors to consider when applying the nitrogen removal cost-effectiveness estimates include the following:

- The social values of reduced nitrogen loadings are unknown. But it appears that the presence of the hypoxic zone, and the evidence that schools of fish avoid it, increases the costs of harvesting fish due to an increase in travel cost. Also, public funding aimed at reducing nitrogen inflows to the Gulf of Mexico suggests that wetland impacts on nitrogen are likely to be of policy interest.

- There is little similarity between hypoxia's effect on the ecosystems of the open and deep waters of the Gulf of Mexico and the effect on the much shallower waters of bays and along shorelines.

Figure 7
**Quantity of nitrogen removed relative to wetland cost in the Upper Mississippi and Ohio River watersheds**




Source: USDA, Economic Research Service analysis of the estimated quantity of nitrogen that a new wetland can remove (figure 4) and the cost of new wetlands (figure 1).

- The cost-effectiveness estimates reflect reduction in nitrogen entering a stream, lake, or river. The estimates do not necessarily reflect the cost effectiveness of reducing downstream impacts because they do not account for in-stream nitrification.

- The costs and the effectiveness of wetlands vary by wetland size. Because of limits to modeling capabilities, this analysis considers only 10-acre wetlands.

## Survival of imperiled species

Research results do not reveal how a change in wetland acreage will affect species' survival probabilities. Also, with only a few estimates of people's willingness to pay for a marginal increase in survival probabilities, we cannot value wetlands' species protection benefits.

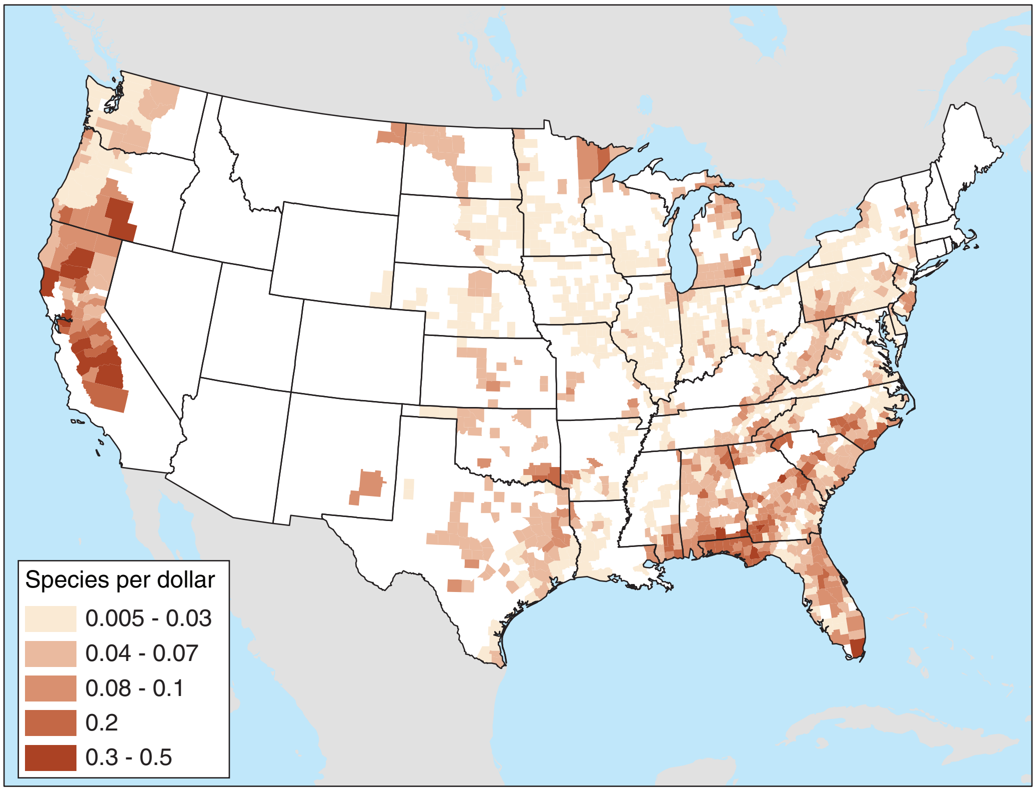
We did examine "hot spots"—areas with large numbers of wetland-associated imperiled species. Dividing by cost, we found species-per-dollar estimates to be greatest in much of California, in southern Oregon, along the Missouri River in the PPR, and in Appalachia (figs. 8a and 8b).

Factors to consider when applying the cost-effectiveness estimates:

- We identify counties with some of the highest numbers of wetland-associated imperiled species and report ratios of the numbers of species relative to wetland costs. These numbers provide a perspective but in no way indicate economic advantages of restoring one wetland relative to another.

Figure 8a

**Number of wetland-associated imperiled* species relative to the cost of new wetlands, by county**




*A classification of “imperiled” is based on various biological and external factors that may affect the long-term viability of a species (Regan et al., 2004).
Note: White indicates that there are either no imperiled species or no cost estimates.
Source: USDA, Economic Research Service using data from NatureServe (2014) and the cost of new wetlands in figure 1.

- The available data identify counties where no or very few wetland-associated imperiled species currently exist, which suggests, but does not indicate, that species-protection benefits might be relatively low.

## Carbon sequestration in the PPR and MAV

The value of the GHG impacts of wetlands throughout the MAV and much of the central and western PPR, the two areas where wetlands’ GHG impacts have been estimated, exceeds our estimate of the cost of restoring and protecting wetlands (fig. 9). If a 5-percent SRD was applied, wetlands’ carbon sequestration benefit-cost ratios would fall by 70 percent. Even under this conservative scenario, benefit-cost ratios are greater than one in the northwestern part of the PPR and throughout much of the MAV.

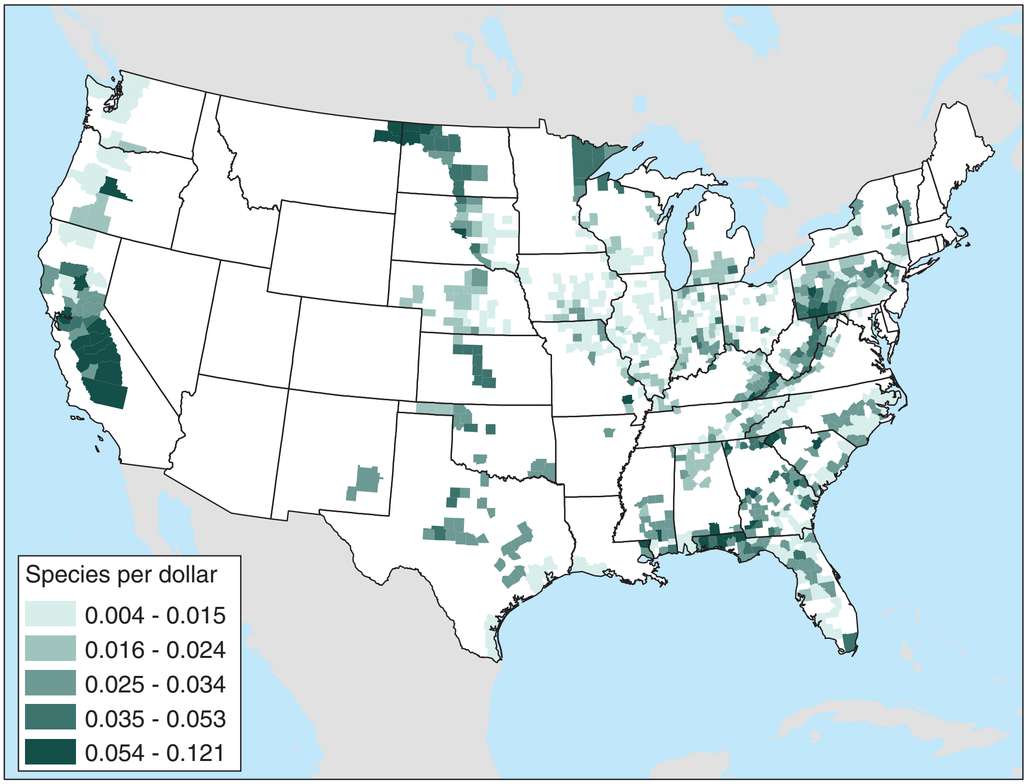
Factors to consider when applying the carbon sequestration benefit-cost estimates:

- Benefit estimates assume that the wetlands are restored on agricultural land.

- Benefit estimates assume that the wetlands are permanent. Converting these wetlands back to agricultural use would result in the release of $CO_2$ and possibly a very substantial fall in benefits.

Figure 8b

**Number of wetland-associated imperiled* vertebrae species** relative to the cost of a new wetland, by county**




*Includes birds, mammals, amphibians, fish, and reptiles. **A classification of "imperiled" is based on various biological and external factors that may affect the long-term viability of a species (Regan et al., 2004).
Note: White indicates that there are either no imperiled vertebrae species or no cost estimates.
Source: USDA, Economic Research Service using data from NatureServe (2014) and the cost of new wetlands in figure 1.

- The economic benefits of marginal changes in GHGs are those prescribed by Executive Order 12866 (U.S. Government, 2010). Agencies are required to use these values to assess the cost and benefits of GHG policy and program options.

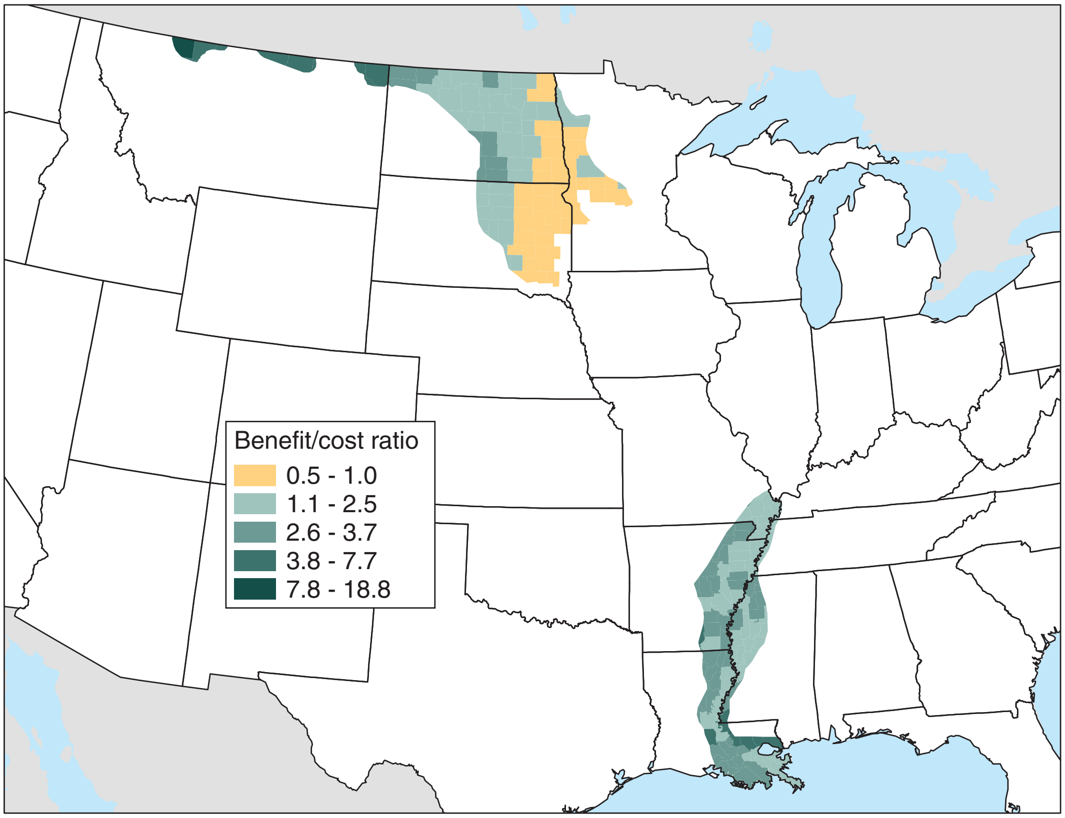
## Considering benefits together

Two of our benefit estimates overlap in the PPR: duck hunting and carbon sequestration. Summing the two PPR benefit-cost ratios, we find that wetlands in the western areas of the PPR have the highest ratios (fig. 10). This is expected given the symmetry in the hunting and GHG cost-benefit ratios.

We estimated nitrogen-removal cost effectiveness in parts of the PPR. In these areas, 2 to 8 pounds of nitrogen are removed per dollar—$0.12 to $0.50 per pound—which is in the lower end of the $0.03 to $7.00 range estimated for wetlands across the Upper Mississippi and Ohio River Valleys. But, being different units of measure (e.g., pounds of nitrogen versus dollar benefits), the values are not additive.

We found no biophysical or economic data or models that would enable us to estimate wetlands' species-protection benefits. Spatial data indicate that the 57 percent of counties in the contiguous 48 States have at least one wetland-associated imperiled species. But restoring a wetland in any county, even in one that currently has no wetland-associated imperiled species, may affect species'

Figure 9
**Wetlands' carbon sequestration benefits relative to wetland cost in the Prairie Pothole Region and the Mississippi Alluvial Valley, by county**




Source: USDA, Economic Research Service analysis of wetlands' impacts on greenhouse gases, (reported by Euliss et al. (2006) and the USFWS (2005)), estimated values of marginal changes in greenhouse gasses (reported by U.S. Government (2010)) and the wetland cost estimates in figure 1.

survival probabilities.[28] Conversely, it is also possible that restoring a wetland in counties with many imperiled species may provide no support. Biophysical models that can predict wetlands' impacts on species' survival probabilities would be very insightful. Additionally, studies that estimate the relative strengths of the public's preferences (or their WTP) to protect different species are limited. As a result, it is difficult, at best, to evaluate the tradeoffs of restoring alternative wetlands.

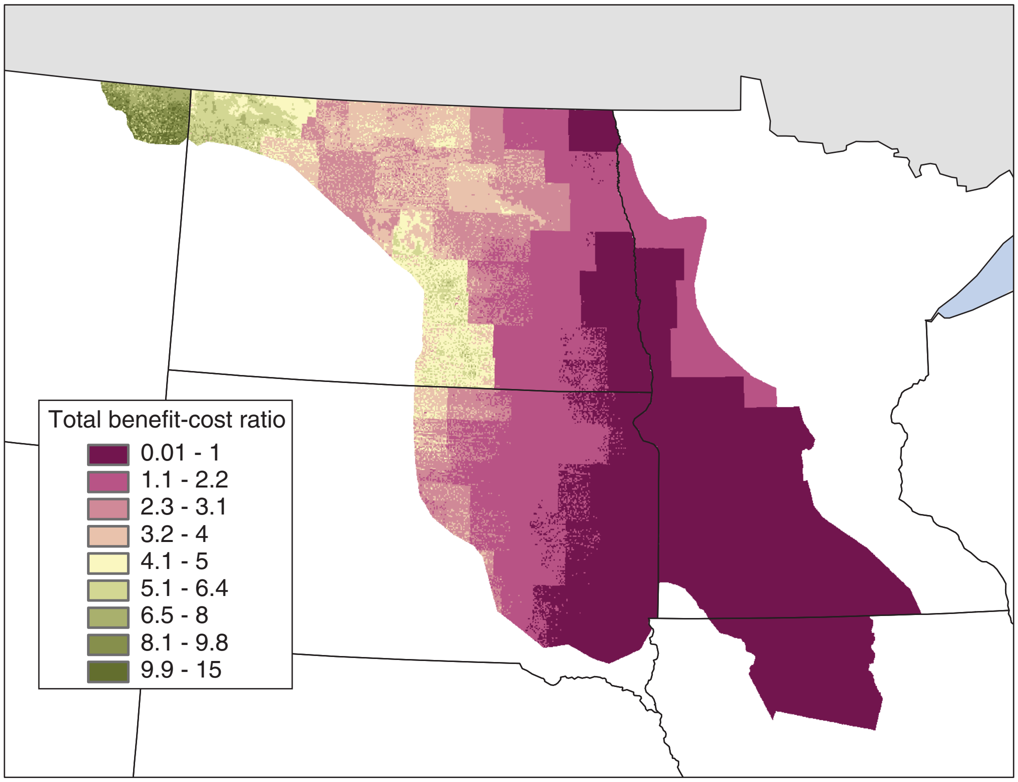
## The current distribution of restored and protected wetland acreage

Although we did not estimate many wetland benefits, a comparison of benefits with the spatial distribution of wetland conservation acreages illustrates how the targeting of wetland conservation efforts might be improved. WRP contract acreage tends to be more concentrated in the MAV, the eastern portion of the PPR, and parts of California and Oregon (fig. 11). We find CRP wetland-related contract acreage spread across much of rural America, but with considerable concentration in the PPR (fig. 12). In a comparison with benefits, the most compelling findings include the following:

[28]Some of these counties may have no wetlands. Some of these counties may have few, if any, wetlands affected by agriculture. There is also the possibility that a wetland can have an indirect effect on a species that is not readily apparent. For example, a wetland's effect on nitrogen loadings could affect the survival rate of a species that is not wetland-associated.

Figure 10

**Sum of duck-hunting and carbon-sequestration benefit-cost ratios of new wetlands in the Prairie Pothole Region**



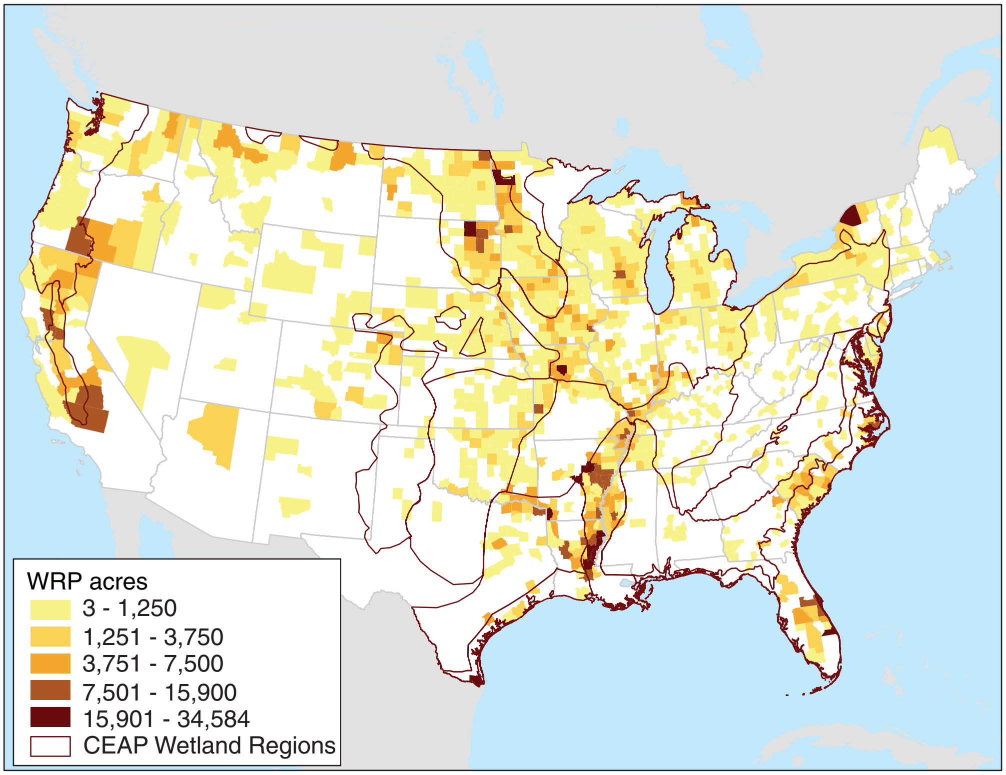

Source: USDA, Economic Research Service analysis of the benefits of duck hunting in figure 3, wetlands' impacts on greenhouse gasses, (reported by Euliss et al. (2006) and the USFWS (2005)), estimated values of marginal changes in greenhouse gasses (reported by U.S. Government (2010)), and the cost of new wetlands in figure 1.

- Figure 11 indicates that, in the PPR, WRP acreage is denser in the eastern portion, where costs tend to be higher (see fig. 2) and duck hunting benefits lower (see fig. 3). Additionally, the total (hunting plus carbon sequestration) benefit-cost ratio estimates are higher where WRP contract density is lower (see fig. 10). This suggests that, within the PPR, targeting wetland restoration and protection funding toward the western areas is likely to provide the greatest net social benefit—note that not all benefits have been estimated. Furthermore, while the imperiled species per-dollar estimates are high in the western portion of the PPR (see fig. 8a), the value of wetlands' species protection is not necessarily higher.

- Figure 12 indicates that in the PPR, the CRP wetland-related contract acreage is denser in the eastern portion, where (as pointed out above) costs tend to be higher (see fig. 2) and the duck hunting benefits lower (see fig. 3). Because the CRP contracts are only 10 to 15 years, carbon sequestration benefits will be substantially less than the values reported here (unless the CRP-funded wetlands remain in place long after contracts expire) (see box "The Value of GHG Impacts Depends on GHG Emissions Over Time" on page 25).

- The MAV has some of the highest density of WRP acreage (see fig. 11). The carbon sequestration benefits of the WRP acreage in most of this area tend to exceed costs—and these wetlands provide other benefits, including biodiversity and water quality. This result provides some justification for the greater density of the WRP acreage.

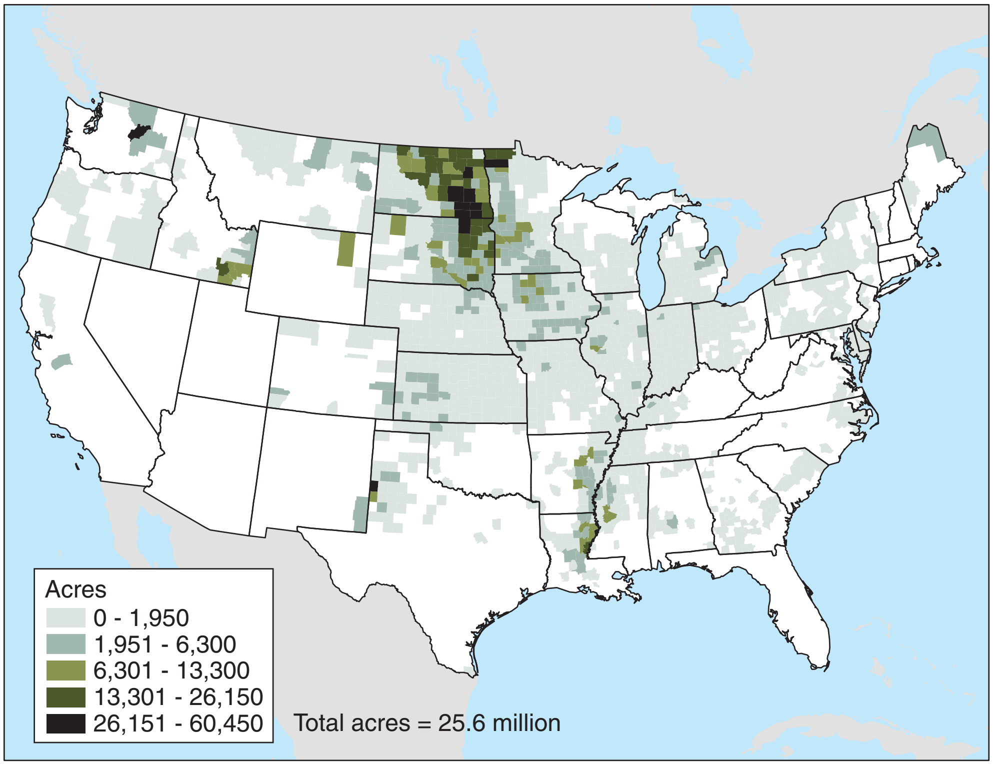
Figure 11
**Number of acres enrolled in the Wetlands Reserve Program, by county**




Source: USDA, Economic Research Service analysis of USDA, Natural Resources Conservation Service confidential data on Wetlands Reserve Program contracts.

Figure 12

**Distribution of Conservation Reserve Program wetland-related contract acreage (both wetland and upland), by county**




Source: USDA, Economic Research Service using confidential program data from USDA, Farm Service Agency, Conservation Reserve Program.

# Conclusions

This study generates spatial estimates of the cost of restoring and preserving wetlands and the social values of some of the benefits they and their ecosystems produce. The results illustrate large spatial variations in wetland costs and benefits and economic advantages to targeting wetland conservation efforts. This information is designed to aid public policymakers, program operators, nongovernmental organizations, and the general public in efforts to employ evidence-based decisionmaking in wetland conservation. While limited by information constraints, this study begins to answer questions such as the following: Where are restored wetlands likely to be most cost effective at removing nitrogen? How much would duck hunters be willing to pay for additional wetlands across the PPR? How valuable are wetlands' greenhouse impacts? What is the likely cost of establishing a new wetland? Answers to these and similar questions are fundamental in constructing and improving conservation contract ranking criteria, such as environmental benefit indexes.

The findings and modeling framework discussed here provide guidance in setting future research priorities in two ways. First, and most apparent, the findings indicate where information is missing and draws attention to where further biophysical and economic research might be most useful. For example, we found that wetlands of the Upper Mississippi and Ohio River watersheds can be, but are not always, very effective at removing nitrogen, but we did not estimate the effectiveness of wetlands elsewhere. This result suggests that analyses of the nitrogen removal rates of wetlands in areas where nitrogen loadings impose social costs could prove to be useful.

Second, the study's analytic framework can help lay out relevant questions. First, is the value of the wetland benefit, $D(g(f(x)))$, likely to be large relative to other benefits? Second, how weak is the weakest link or what link(s) is(are) missing: The data on wetland characteristics: $x$? The models of wetland service: $f(x)$? The models of amenities produced: $g(f(x)))$? Or the demand or willingness to pay function: $D(g(f(x)))$? And third, how can the derivation of one model help narrow the focus of research on the other models? For example, before allocating research funding to estimate wetland effects on groundwater recharge, consider the economic importance of groundwater: Do many people in the area consume groundwater either directly or through irrigation applications? Will additional groundwater reduce the potential for seasonal shortages of stream flows? Also, consider other critical biophysical links: Are the affected aquifers the same as those that feed consumers and streams? If the answers to these questions are "no," then research efforts might be better channeled elsewhere.

After considering the contribution that economics can make and identifying some key information gaps, we offer suggestions for future research. This list is not comprehensive; rather, it is meant to illustrate areas where additional estimates will be useful.

- Duck hunting benefits were found to be high but the estimates drew on willingness-to-pay analyses that are dated. Because of the economic significance of duck hunting benefits, accuracy is important. An update of individuals' willingness to pay for improved hunting quality can help solidify results but will require data collection and analysis.

- Wetlands' duck hunting benefits were found to be high and variable, but the spatial extent of the biophysical models are limited. This suggests that research that expands the spatial extent of biophysical models might be useful.

- Dollar-per-ton estimates are available for some soil conservation benefits (Hansen and Ribaudo, 2008), and these estimates can be used to value wetlands' soil conservation benefits.[29] But, to do so, estimates of wetlands' effects on erosion are needed. One can expect erosion rates on converted wetland acreage to be less than county average rates because wetlands are commonly restored on flat lands. Land-use data, such as the National Resources Inventory, might be used to estimate erosion rates.

- The results of this study suggest that, in some areas, wetlands can be cost effective at removing nitrogen from nearby waterways. Thus, additional analyses of nitrogen removal by other sizes and types of wetlands across different-sized watersheds within and outside of the study area may be useful.

- We did not review literature on public consumption of groundwater—we did not have estimates of wetlands' effects on groundwater. Economic research that provides spatial estimates of the public's willingness to pay for increased groundwater will indicate to hydrologists where they might focus analyses of wetlands' effects on groundwater. Similarly, if research by hydrologists identifies areas where wetlands do affect groundwater, economists will see where to focus research on the valuation of groundwater.

- Similarly, economic research that generates spatial estimates of the social cost of algae blooms in freshwaters (due to phosphorus loadings) can suggest to biophysical scientists areas of the country where they might focus research on phosphorus removal by wetlands.

- The public's willingness to pay to protect species appears to be high but not infinite, as indicated by historical expenditures (Shogren et al., 1998). A further review of the economics literature along with a survey of peoples' preferences across species and levels of protection (e.g., changes in survival probabilities) can provide biologists with insights on those species that are of greatest public interest, hence, areas where they might prioritize research. The same information can also give those engaged in conservation efforts a better idea of how to prioritize funding for species protection. There are data that might enable analysts to examine the biophysical effects of wetlands on imperiled species. For example, information on the effects of habitat on species population may be contained within various descriptive text fields in NatureServe's central database (www.fort.usgs.gov/Products/ProdSpecies_Index.asp) and on the U.S. Geological Survey Web site (www.fort.usgs.gov/Products/ProdSpecies_Index.asp).

[29]The soil-productivity benefit estimates reported in Hansen and Ribaudo (2008) should not be applied when land is not to be moved back into agricultural production. In cases where the land is idled temporarily, as with the CRP, one must account for the delay in the yield gains resulting from the increase in productivity.

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# Appendix 1—The Costs of Wetland Conservation Options

Our primary data are from a Wetlands Reserve Program (WRP) contract file. These data provide our associated variable: the total cost of a wetland (*TWC)*. TWC is the sum of the restoration and easement costs of each contract. WRP contract data are agreed-upon contract payments and acreages and estimates of the wetland restoration costs—actual costs may differ. Overall, we expect these data to be representative of WRP costs. As long as the contract data are not systematically biased, they should be suitable for estimating wetland cost models. The data include a variable that identifies the county where the contract is located. We use this variable to link the WRP data to other data. We specified our cost model as:

*TWC = f(Acres, AcresSq, AgrValue, AgrValueSq, Rural)*

Information on the size of each contract (*Acres*) is included in the WRP contract file. As discussed earlier, we expect the cost of restoring wetlands to increase with the size of the wetland, all else being equal. And we expect this relationship to be nonlinear; hence, we have included the second-degree term acres squared (*AcresSq*). We also expect the nuisance costs of wetlands to be dependent on the size of the wetland and that the effect is nonlinear, all else being equal. We estimate each cost model using observations from the same wetland region to minimize variations in other determinants of restoration costs.

We have no county-level data on farmland values. But per-acre farmland rental rate data are available and can serve as a suitable proxy for variation in the value of the land's agricultural productivity. We generated indicators of the agricultural value of land in contracts (*AgrValue*), by multiplying the contract acreage (*Acres*) by the county-average dry land rental rates reported by USDA's National Agricultural Statistics Service (NASS). NASS generates rental values from various sources, including the recently initiated county-level cash rents survey (USDA, NASS, 2009). Because we expect a nonlinear relationship, we include the square of *AgrValue* (*AgrValueSq*).

We use the Population-Interaction Zones for Agriculture (PIZA) data to determine locations that may have urban pressure (*Rural*). The PIZA scores are designed to measure unban influence. The PIZA data classify counties as rural or as one of three levels of urban. (Documentation and PIZA data are available at: www.ers.usda.gov/data-products/population-interaction-zones-for-agriculture-(piza).aspx). We use the PIZA scores to generate a zero-one dummy variable, Rural, that equals one when urban influence is likely to be minimal.

Data that quantify actions taken to restore wetland hydrology and ecosystems and eliminate or control invasive species are not available. To overcome or minimize the effect of this shortcoming, we estimated separate cost functions for each of the USDA, Natural Resources Conservation Service's Conservation Effects Assessment Project (CEAP) wetland regions (see fig. 1). By design, CEAP regions are defined by wetland similarities. It seems reasonable to assume that, within a wetland region, determinants of the cost of restoring a wetland—other than the size of the wetland—are less likely to vary than are determinants across wetlands regions.

We acknowledge that *AgrValue* and Acres are correlated, as the size of a contract is one of the drivers of the cost of land and the cost of restoring and improving a wetland. This correlation can weaken conclusions one can draw about the signs, sizes, and statistical significance (due to inflated standard errors) of the estimated coefficients. However, correlation between independent variables is a model structural issue: it affects one's ability to assess causality. It does not introduce systematic bias and therefore may not weaken the model's prediction capabilities, which is our motivation for estimating the cost functions.

## Results and Model Applications

The adjusted R-squares of the estimated models top out at 0.88. Half capture more than 75 percent of the variation in the dependent variable; the three weakest capture 53 to 60 percent (table A1). Given the complications resulting from variable correlations, one cannot make any conclusions about the economic implications of the estimated coefficients.

Before generating estimates of expected TWC for a new wetland, we need to identify appropriately representative contract sizes (Acres) for each wetland region. A comparison of the means and median indicates that the distribution of contract sizes is skewed toward the larger sizes (table A1). We chose to set Acres equal to the median-sized contract (for the region) as a useful representative measure since half of the contracts will be greater in size and half will be smaller. Because the cost functions are nonlinear, one does not know how cost estimates based on mean values compare with the potential range of estimated costs. For example, using the average wetland size is not likely to generate an estimate of average cost. One advantage of using median values is that the models will generate median cost estimates when based on median wetland sizes.

We generated estimates of total and average wetland costs for each county. Estimates show that there is less variability in projected costs in much of the Southeast than elsewhere; hence, benefit-cost ratios are less likely to be driven by variation in costs in the Southeast (see fig. 2). The Appalachian Highlands—with a cost model that captures 88 percent of the variation in the dependent variable—has the widest range in predicted wetland costs, with values in a scattered distribution across the area. Predicted costs also vary widely across the Prairie Pothole Region, but the values show a consistent pattern of increase as one goes from the region's northwest counties to the southeastern counties.

Table A1

**Estimated coefficients of the wetland cost models of the 10 wetland regions**

| Wetland region | Adjusted R-square | Variable* | Coefficient | Standard error** | Number of observations |
|---|---|---|---|---|---|
| Glaciated Interior Plains | 0.81 | Intercept | -1247.23 | 3162.32167 | 3,656 |
| | | AgrValue | 13.323 | 0.58039 | |
| | | AgrValueSq | -2.98E-07 | 9.85E-07 | |
| | | Acres | 434.2865 | 38.52275 | |
| | | AcresSq | -0.02602 | 0.00738 | |
| Pacific Mountain | 0.53 | Intercept | -31659 | 27684 | 265 |
| | | AgrValue | 46.83228 | 4.69902 | |
| | | AgrValueSq | 0.000506 | -0.00004666 | |
| | | Acres | -419.998 | 175.86181 | |
| | | AcresSq | 0.37051 | 0.0653 | |
| Prairie Pothole Region | 0.79 | Intercept | -3870.37 | 2615.75999 | 2,468 |
| | | AgrValue | 29.22938 | 0.57392 | |
| | | AgrValueSq | -0.00016 | 0.00000509 | |
| | | Acres | -421.041 | 26.21381 | |
| | | AcresSq | 0.19056 | 0.00684 | |
| Gulf Atlantic Coastal Flats | 0.58 | Intercept | 207399 | 71840 | 283 |
| | | AgrValue | -3.57109 | 1.7062 | |
| | | AgrValueSq | 5.86E-06 | 9.89E-07 | |
| | | Acres | 790.5241 | 102.37909 | |
| | | AcresSq | -0.02619 | 0.00321 | |
| Appalachian Highlands | 0.88 | Intercept | 6509.232 | 1167.2668 | 1,537 |
| | | AgrValue | 40.12727 | 1.6565 | |
| | | AgrValueSq | -0.00016 | 0.00003148 | |
| | | Acres | -422.888 | 60.72724 | |
| | | AcresSq | 0.23552 | 0.07277 | |
| High Plains | 0.71 | Intercept | 14978 | 8847.30368 | 240 |
| | | AgrValue | 12.08406 | 3.76541 | |
| | | AgrValueSq | 0.000264 | 0.00009641 | |
| | | Acres | 473.1165 | 97.13946 | |
| | | AcresSq | -0.31101 | 0.05205 | |
| Mississippi Alluvial Valley | 0.60 | Intercept | 4339.446 | 9551.82475 | 1,417 |
| | | AgrValue | 23.84496 | 2.49292 | |
| | | AgrValueSq | -5.7E-05 | 0.00001686 | |
| | | Acres | -417.548 | 108.46545 | |
| | | AcresSq | 0.14741 | 0.02848 | |
| Central Valley | 0.76 | Intercept | -69276 | 44584 | 199 |
| | | AgrValue | 15.05214 | 9.40191 | |
| | | AgrValueSq | 9.53E-05 | 0.00012050 | |
| | | Acres | 1597.231 | 214.96889 | |
| | | AcresSq | -0.26999 | 0.07019 | |

Continued—

Table A1

**Estimated coefficients of the wetland cost models of the 10 wetland regions—continued**

| Wetland region | Adjusted R-square | Variable* | Coefficient | Standard error** | Number of observations |
|---|---|---|---|---|---|
| Gulf Atlantic Coastal Plains | 0.70 | Intercept | 11896 | 11909 | 963 |
| | | AgrValue | 8.59126 | 1.62986 | |
| | | AgrValueSq | 2.86E-05 | 0.00000621 | |
| | | Acres | 429.924 | 47.14001 | |
| | | AcresSq | -0.03638 | 0.00344 | |
| Central Plains | 0.84 | Intercept | 10650 | 4277.74641 | 706 |
| | | AgrValue | -3.57894 | 1.2879 | |
| | | AgrValueSq | 0.000118 | 0.00000549 | |
| | | Acres | 712.5908 | 43.61911 | |
| | | AcresSq | -0.11722 | 0.0055 | |

*In the first regression runs, we found that, in most cases, the coefficient on Rural was insignificant. Hence we dropped the variable. The R-square fell by less than two points.

**Despite the fact that our variables are likely to be correlated and, consequently, the standard errors of the coefficients inflated, nearly all variable coefficients are statistically significant at the 99-percent level.

Source: USDA, Economic Research Service.

# Appendix 2—Conceptual Framework for Assessing Environmental Benefits

## Economic and biophysical frameworks

The productivity of wetlands varies similar to the productivity of agricultural lands. There are different types of wetlands and different types of agricultural lands. Their outputs are affected by a variety of environmental factors. On the agricultural side, markets bring society's preferences and production resources into alignment: producers choose what, how, and how much to produce based on prices and factor productivity. The prices consumers accept reveal society's preferences.[30]

On the wetland side, there are few markets that can provide measures of the values of wetland services. Instead, efforts to cost effectively restore and protect wetlands must rely on estimates of wetland productivity and the public's demands.

This appendix provides a structural framework suitable to evaluating and comparing wetland conservation benefits. Consider the following simple algebraic formulation using $x$, $f(x)$, $g(f(x))$, and $D(g(f(x)))$.

- $x$: measures of eco-environmental characteristics, such as the size and type of wetland, daily temperature and precipitation, climatic conditions, surrounding land uses, quantity and quality of inflow waters, and the existence of invasive species.

- $f_i(x)$: the quantity of service $i$ that a wetland produces. This will be a function of eco-environmental characteristics. For example, changes in coastal wetland conditions affect offshore fishery populations.

    - $g_j(fi(x))$: the quantity of amenity $j$ that is provided by a wetland. That is, a single service, $f_i(x)$, can produce a variety of amenities. For example, a new wetland can yield an increase in fish populations, $f_{FISH}(x)$, and the increase in fish populations can improve the catch of commercial fishing operations ($g_{HARVEST}(f_{FISH}(x))$) and the quality of recreational fishing ($g_{FISHINGQUALITY(fFISH}(x))$).[31]

- $D_{k(}g_{j(}f_{i(x))):}$ an amenity can be of value to people and commercial operations. When amenities directly affect commercial markets, the measurement of value is straightforward—the value of a change in the amenity is equal to the resulting change in profits, $D_{PROFITS}(g_{HARVEST}(f_{FISH}(x)))$. For example, extending the fish example, one could estimate the economic impact on commercial fishing by multiplying the dockside price of fish by the gain in harvest. When amenities affect nonmarket goods, estimating willingness to pay is more complicated. Again, extending the fish example, demand for improved fishing quality can be measured through a population survey where individuals are asked what they would be willing to pay for improved fishing quality. Or, in situations where participants are choosing from multiple sites with differences in fishing

[30]When market failures exist (such as when farmers do not pay for damages or are not compensated for benefits their practices generate), market equilibrium is not likely to be socially optimal.

[31]Because the quality of recreational fishing is likely to depend on both catch rates and the size of fish caught, developing a measure of such an amenity, $g_{FISHINGQUALITY}(f_{FISH}(x))$, is challenging.

quality, willingness to pay can be derived from data on travel costs, fishing quality, and other environmental and population characteristics.

The boundaries between *x, f(x), g(f(x)), and D(g(f(x)))* are "grey." For example, one could define a wetland's effect on duckling or adult populations as a service. We specify the link between a wetland and its amenities in a two-step process, *g(f(x))* to emphasize that there can be a chain of relationships involved—that is, a new wetland removes nitrogen, which reduces nitrogen in river systems, which reduces nitrogen entering estuaries, which reduces hypoxia, which affects a bay's ecosystem, which affects marine fisheries.

The specifications of *x*, *f(x)*, *g(f(x))*, and *D(g(f(x)))* are subject to the data and models that are available. Also, past environmental benefit analyses have applied shortcuts to measuring these relationships.

## Alternative approaches for benefits analyses

### *Reduced form models*

Reduced form models commonly shortcut the need to estimate the effect of an environmental service on an amenity by modeling demand as a function of the service *D((f(x))*. A variety of studies estimated regional and national water quality benefits of soil conservation practices by, first, using the Revised Universal Soil Loss Equation to estimate erosion, *f(x)*, and, second, estimating willingness to pay for changes in water quality as a function of soil erosion (Sullivan et al., 2004; Hansen et al., 2002; Feather et al., 1999). In a regional analysis of the Conservation Reserve Program's hunting benefits, Hansen et al. (1999) estimated a reduced-form spatial model where five of the independent variables (the surrounding mix of agricultural land uses) served as a proxy for the environmental amenity (hunting quality).

### *Proxy variables*

Proxy or indicator variables can sometimes be substituted for variables or data that are not available. For example, one could use the observed percent change in fish catch rates in the fisheries industry as a proxy for the percent change in the quality of recreational fishing. The strength of a proxy variable hinges on the correlation between the proxy variable and the true variable's value.

### *Meta-analysis*

Meta-analyses use findings and models reported in previous research as data. Meta-analyses have been used to directly value wetlands, sidestepping the need to estimate the many biophysical relationships and economic values of the amenities. A major shortcoming of meta analyses related to wetlands is that, in most cases, they provide a single national or a limited number of regional values, and, hence, capture virtually no spatially resolution to variations in wetland benefits. Another shortcoming of meta analyses is that the studies used have not evaluated wetlands that are representative of the kinds of wetlands that are restored on agricultural land. For example, Brander et al. (2006) used the results of a study by Cooper and Loomis (1993) in a meta analysis of wetland values. But Cooper and Loomis used a travel cost analysis to determine hunters' willingness to pay to hunt sites on public lands. Additionally, willingness to pay is determined by the sizes and proximities of surrounding populations, water levels in the wetlands, and other factors—these factors and features of public wetlands are not likely to be representative of wetlands restored on agricultural lands.

Borisova-Kidder (2006) attempted to estimate a wetland-benefits model suitable for U.S. agriculture policy and program analyses but was unable to control for differences in wetland benefits across wetland types. For example, there was only one appropriate study that evaluated a wetland in the Prairie Pothole Region of the Upper Midwest.

# Appendix 3—Predicting New Wetlands Effects on Ducks

We developed models that predict the number of hatchlings a new wetland is likely to produce for the five dominant duck species of the Prairie Pothole Region (PPRP (blue-winged teal (A*nas discors*), gadwall (*A. strepera*), mallard (*A. platyrhynchos*), northern pintail (*A. acuta*), and northern shoveler (*A. clypeata*)). Put simply, we model hatchlings (*H*) produced by a specific species at a wetland as a function of the nesting pairs (*NP*) the wetland attracts, nest success (*NS*), re-nesting propensity (*NI*), and clutch size (*CS*):

$H = NP * NS * NI * CS.$

We built this relationship from data on duck species, findings of prior research, and two types of models—a nesting pair model and a nest success model—that were developed for the PPR of North and South Dakota and northeastern Montana. We extended this model and drew on other resources to generate estimates of wetland effects on hatchlings in the PPR of Iowa and Minnesota.

We generated estimates of *NP* (nesting pairs) from a set of "pair-wetland relationship models," one for each species and wetland class (i.e., temporary, seasonal, semi-permanent, lake, river (Steward and Kantrud, 1971, 1972; Cowardin et al., 1995) that were developed for field-level evaluations of the expected effects of a new wetland. Pair models were developed from breeding pair and wetland condition data collected during the U.S. Fish and Wildlife Service Annual Breeding Waterfowl Survey from 1987-2008 (Cowardin et al., 1995, Reynolds et al., 2006). We applied these pair models across 1 mile x 1 mile fishnet cells. This enabled us to generate a GIS surface of the number of breeding pairs added with a new wetland.

Nest success (*NS*) is the probability an egg survives all exposure days (Mayfield, 1975) and is a primary driver of duck population maintenance (Hoekman et al., 2002). Nest success is positively related to the share of surrounding grassland cover (Reynolds et al., 2001; Stephens et al., 2008). To estimate *NS*, we began with available daily nest survival rate (DNS) models (Mayfield, 1975) and developed spatially explicit models from Reynolds et al. (2001). The DNS model for a species $_i$ is expressed as $DNS_i = a_i + (b_1 * \textit{percent grass}) + (b^2 * \textit{easting}) + (b^3 * \textit{northing}) + (b_4 * \textit{easting} * \textit{northing})$, where $a_i$ is the intercept, *percent grass* is the share of grass cover within a 1-mile radius of the centroid point of each pixel, and easting and northing are longitudinal and latitudinal measures of the centroid point (Reynolds et al., 2001).[32]

We used spatially specific measures of *percent grass f*rom landcover data from Thematic Mapper satellite imagery (93.5 ft. x 93.5 ft. pixel) for the years 2001-03 (Habitat and Population Evaluation Team, USFWS Region 6, Bismarck, ND, unpublished data). With these data and employing a moving window analysis, we calculated *percent grass* (which includes CRP) within the surrounding 3.14-mile$^2$ area (1 mile radius) of each pixel.

Finally, to generate spatial estimates of *NS* (the probability that a nest survives all days from when the first egg is laid to the day when the eggs hatch) for each species, we raised the *DNS* estimate of each species to the power of the number of nest-exposure days.

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[32]Values of *easting and northing* are expressed in meters and are based on the Universal Transverse Mercator, NAD 83, zone 14. In our analysis, easting and northing were both divided by 100,000.

Nesting propensity (*NI*) accounts for cases where a nesting pair re-nests after losing its nest. By re-nesting, the nesting pair can still add to the hatchling population so it must be accounted for. Nesting propensity of a given species was treated as a constant and was taken for Bellrose (1980).

Average clutch size (*CS*) varies by species: mallard=8.4, gadwall=9.9, teal=10.2, pintail=9.9, and shoveler=7.1.

We generated estimates of the effect of new wetlands across the PPR and found that productivity ranges from near 0 to 57 hatchlings per acre. The distribution of the productivity of wetlands is skewed toward the lower end, with the bottom 10 percent of the wetlands likely to produce 3 hatchlings per acre or less. The productivity of the top 10 percent is 18 hatchlings per acre or more. The median level of productivity is 5.5 hatchlings per acre.

# Appendix 4—Calculating Wetlands' Nitrogen Removal Potential

We estimated the potential nitrate mass removal by wetlands as the product of nitrate mass load intercepted and the nitrate removal rate for locations across the Upper Mississippi and Ohio River Basins. Detailed descriptions on the estimation and application of the models that generated the mass loads and nitrate removal rates and supporting data sources are available in Crumpton et al. (2006) and Crumpton et al. (2010). In summary, Crumpton et al. (2006) generated spatial estimates of nitrate mass load based on the average water flow that would be intercepted by a wetland and the average nitrate concentration in that flow. Percent nitrate removal was calculated as a function of average hydraulic loading rate expected for wetlands at each location. Crumpton et al. (2010) extended the Crumpton et al. (2006) estimates to include the effect of tile drainage on nitrate concentration. Essentially, they estimated field nitrogen mass loss within a wetland watershed as a function of land uses (fraction of area in corn and soybean cultivation), tile drainage (fraction of crop land with tile drainage), and hydraulic loading rate and found all variables to be statistically significant (Crumpton et al., 2010).

Our nitrogen-removal estimates were generated using the above models, Landsat land cover data from Landsat Thematic Mapper imagery, NRCS Soil Survey Geographical (SSURGO) data, 1980-2005 U.S. Geological Survey stream gauge data, and 20-year simulations to capture expected variations in precipitation and rainfall intensity and their subsequent impact on nitrogen movement and wetland effectiveness. We assumed that wetlands are strategically placed at the bottoms of 1,000-acre wetland watersheds.33 Our review of the literature and the analyses supporting the models we have applied indicate that a 1,000-acre wetland watershed is likely to be large enough to supply enough water to maintain a healthy wetland ecosystem and not atypical within our study areas. Furthermore, a recent study found that, in northwestern Ohio, a wetland of roughly 10 acres is most cost effective at nitrogen removal (Hansen et al., 2012). While other sized wetlands and wetland watersheds might have been considered, for the sake of brevity, we have not done so here. Our results indicate that, depending on where it is located, a new 10-acre wetland may remove 11 pounds to nearly 1 ton of nitrogen per wetland acre (see fig. 4).

[33]Strategic locations are locations where wetlands will capture surface and subsurface water flows from fields, as opposed to placing wetlands indiscriminately. When it comes to wetland location, the N removal estimates reported here should be viewed as upper-bound estimates.