Agricultural conservation practices increase wetland ecosystem services in the Glaciated Interior Plains

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Abstract. The Glaciated Interior Plains historically supported a broad variety of wetland types, but wetland losses, primarily due to agricultural drainage, range from 50% to 90% of presettlement area. Wholesale land use change has created one of the most productive agricultural regions on earth, but wetland conversion has also led to the loss of the ecosystem services they provide, particularly water quality improvement, flood de-synchronization, carbon sequestration, and support of wetland-dependent species (biodiversity). Nearly three-quarters of the Glaciated Interior Plains fall within the Mississippi River drainage basin, where the combination of extensive tile drainage and fertilizer use has produced watersheds that contribute some of the highest nitrogen yields per acre to the Mississippi River. Wetland conservation practices implemented under Farm Bill conservation programs have established or involved management of nearly 110 000 ha of wetlands, riparian zones, and associated ecosystem services over the period 2000–2007. We estimated the cumulative ability of these conservation practices to retain sediment, nitrogen, and phosphorus in Upper Mississippi River Basin watersheds. Estimated retention amounts to 1.0%, 1.5%, and 0.8% of the total N, sediment, and P, respectively, reaching the Gulf of Mexico each year. If nutrient reduction is estimated based on the quantity of nutrients exported from the Glaciated Interior Plains region only, the numbers increase to 6.8% of N, 4.9% of P, and 11.5% of sediment generated in the region annually. On a watershed basis, the correlation between the area of wetland conservation practices implemented and per-hectare nutrient yield was 0.81, suggesting that, for water quality improvement, conservation practices are successfully targeting watersheds that are among the most degraded. The provision of other ecosystem services such as C sequestration and biodiversity is less well studied. At best, implementation of wetland and riparian conservation practices in agricultural landscapes results in improved environmental quality and human health, and strengthens the rationale for expanding conservation practices and programs on agricultural lands.

Key words: carbon sequestration; flood abatement; Glaciated Interior Plains, USA; nitrogen; U.S. Midwest; water quality improvement.

INTRODUCTION

The Glaciated Interior Plains (GIP) covers most of a seven-state area stretching from Ohio to Minnesota, USA, encompassing a broad variety of landscapes and associated wetlands. Sometimes referred to as the “corn belt,” this area is one of the most productive agricultural regions on earth, accounting for >50% of the nation’s corn production (Power et al. 1998), as well as extensive production of soybeans, wheat, and other grains. The wholesale land conversion that occurred to make way for agriculture resulted in a loss of native ecosystems, including wetlands, reducing their ability to provide critical ecosystem services. Globally, wetlands deliver a wide range of services (e.g., water purification, climate regulation, flood regulation, coastal protection, water supply, fish and fiber production, recreational opportunities, and tourism) that contribute substantially to human well-being (Millenium Ecosystem Assessment 2005). Within the GIP, wetland ecosystem services such as flood abatement, water quality improvement, and the support of biodiversity have declined dramatically due to extensive wetland drainage (Hey 2002, Zedler 2003b). One consequence of wetland loss, combined with regionally intense agricultural inputs (fertilizer, pesticides), is the chronic degradation of water quality, particularly in the upper Mississippi River Basin, which ultimately contributes to hypoxia in the Gulf of Mexico. This paper describes what is known about the types and status of wetlands in the GIP and how conservation practices implemented through Farm Bill conservation programs contribute to the restoration of wetland ecosystem services, particularly water quality improvement.

DIVERSITY OF WETLANDS IN THE GLACIATED INTERIOR PLAINS

The GIP of the Midwest United States (Fig. 1) contain a variety of wetland types that exhibit extensive spatial
variability, differing from east to west as precipitation decreases and from north to south with increasing temperatures. The climate of the region is characterized as temperate continental with subhumid to humid summers and cold winters (Trewartha 1981). Mean annual temperatures range from 11.1°C in the east (Columbus, Ohio) to 9.6°C in the west (Des Moines, Iowa), with associated rainfall ranging from 96.2 cm to 77.7 cm (Amon et al. 2002). Much of the region was shaped in the most recent, Wisconsin glaciation, which produced the current distribution and type of wetlands, including depressional marshes, shrub and forested bogs, fens, forested wetlands, and wet meadows. Wetlands of the Midwest can be distinguished on the basis of the dominant water source, consisting of those that receive water mostly from surface flooding (floodplain and riparian forests), precipitation (depressional wetlands, vernal pools, bogs), groundwater (fens, seeps), and a combination of sources (e.g., wet meadows) (Fig. 2). In the east (Ohio, Michigan, Indiana), forested wetlands (depressional, seep, and riparian), marshes, and fens are predominant. Forested riparian wetlands are common along floodplains (Brown and Peterson 1983, Rust and Mitsch 1984, Polit and Brown 1996, Baker and Wiley 2004). These wetlands provide important ecosystem services, including water quality maintenance, habitat, and flood de-synchronization (Table 1).

Depressional wetlands exist on broad, upland flats in topographic low spots, and to a lesser extent, on floodplains and are dominated by either forested or herbaceous emergent vegetation (Galatowitsch et al. 2000, Craft et al. 2007). Depressional wetlands lack strong surface water connections to other aquatic ecosystems and, although they are sinks for pollutants in the local landscape, their role in regional water quality maintenance is less well understood (Table 1). However, they are important habitat for wetland-dependent avian, fish, and amphibian species, and as watering holes for animals. Vernal pools represent a special type of depressional wetland. They are typically smaller in size and forested, with a shorter hydroperiod than depressional wetlands such as marshes (Zedler 2003a). Because of the short hydroperiod, these wetlands lack predators such as fish, making them critical sites for amphibian reproduction (Table 1; Gibbons 2003).

Fens and seeps occur in areas where mineral-rich groundwater discharges. They are typically low in nutrients, particularly P, and high in carbonates with high plant species diversity (Bridgham et al. 1996, Amon et al. 2002). Fens and seeps often have strong connections to regional groundwater flows that deliver carbonates (CaCO₃), as well as nutrients (NO₃⁻) leached from agricultural lands (Amon et al. 2002, Craft et al. 2007).

Wetlands of the central portion of the Midwest (Illinois, Iowa, southern Wisconsin), where annual precipitation is less than in the east, consist of forested floodplains and stream corridors, freshwater marshes, and wet meadows and prairies. Wet meadows possess shorter hydroperiods than marshes in the region. Soils tend to be saturated, and not inundated (Prince 1997). In the eastern part of the region, wet meadows are found in association with oak savannas of the mesic prairie landscape (Nuzzo 1986). These wetlands support a rich diversity of plants (Table 1), and threatened grassland birds such as Greater Prairie-chicken, Tyranneus cupido, that use these habitats for roosting (Toepfer and Eng 1988).

In the north (Wisconsin, Minnesota, northern Michigan), peat-accumulating wetlands (bogs and fens) are common landscape features. Bogs receive essentially all of their water and nutrients from precipitation, making their peat acidic (pH < 4), low in available nutrients, with an enormous store of carbon (Table 1; Grigal 1991, Crum 1995, Bridgham et al. 2001). Other wetlands in the northwestern portion of the region include glacially formed lakes with extensive littoral zones dominated by marsh vegetation (Tiner 2003).

**Wetland Losses, Landscape Change, and Ecosystem Services**

The conversion of wetlands to crop and pasture lands over the past 250 years has transformed the landscape of the GIP. Humans cleared forests, broke sod, and drained wetlands to clear land and facilitate farming. An unintended consequence of this land conversion was degraded water quality, flood damage, diminished biodiversity, and radically altered regional hydrology (Prince 1997, Hey et al. 2005). Because hydrology determines the location of wetlands and their structural and functional properties, agricultural expansion has caused not only an enormous loss of wetland acreage, but has also compromised the ability of existing wetlands to provide their characteristic ecosystem services (Zedler 2003b).

Agricultural land use dominates the region, ranging from 93% agricultural land use in the state of Iowa to 30% in Michigan. Much of this land is drained using drain tile to move water (and associated chemicals) quickly from the field to adjacent ditches and streams. The extent of tile drainage underlying croplands ranges from 50% in Ohio and Indiana (~3 × 10⁶ ha in each state) to 10% (0.9 × 10⁶ ha) in Wisconsin (Mitsch et al. 1999). Over 40% of the N fertilizer used in the United States is applied to agricultural lands in the GIP. Of this, 35% is estimated to run off into receiving waters (Howarth et al. 2002, Zedler 2003b).

As part of these land use changes, large areas of wetlands once common throughout the Midwestern states have been lost, and with them the landscape’s ability to regulate water movement and biogeochemical cycles. Wetland conversion to make way for agriculture came early; for example, the four states that make up the bulk of the GIP lost between 80% and 90% of their original wetland acreage between 1780 and 1980 (Dahl 1990), including Ohio (90%), Indiana (87%), Illinois (85%), and Iowa (89%). In addition, two enormous
wetland complexes, the Great Black Swamp in northwestern Ohio and the Great Kankakee Marsh in northern Indiana and Illinois, each more than 1,000,000 acres (400,000 ha) in size, were systematically drained for agriculture and no longer exist today (Mitsch and Gosselink 2000). Losses in the northern-tier states are lower, with nearly half of the original wetland acreage lost (Michigan lost 50%, Wisconsin 46%, and Minnesota 42%; Dahl 1990). The push for wetland drainage slowed dramatically in Wisconsin and Michigan in the early 1900s, when many peat soils were found to be acidic and deficient in essential plant nutrients (Prince 1997). Collectively, the seven states that make up the GIP have had ~18.6 million ha of farmland drained for agriculture (Mitsch et al. 1999).

Water quality impacts

The loss of wetlands from a landscape removes their capacity to act as sinks and transformers of sediments,
Agricultural sources are responsible for an estimated 58% of the high N export in the Mississippi River (Rabalais et al. 2002). The environmental issues, most notably the hypoxic zone (or the “dead zone”) in the Gulf of Mexico have increased nearly three-fold since 1950 (Hey 2002). Like much of the Glaciated Interior Plains, nearly 90% of the original wetland area in the Midwest (USA) has been drained. Over the past 50 years, fertilizer use in the region has intensified, and N losses have contributed to reductions in water quality in the region. The combination of extensive drainage and intensive row crop fertilization in the GIP has produced watersheds that contribute some of the highest nitrogen yields per acre to the Mississippi River (nearly three-quarters of the GIP region is located in the Mississippi River watershed; Goolsby et al. 1999). The environmental consequences of this are particularly well documented, where high N export has led to downstream water quality issues, most notably the hypoxic zone (or the “dead zone”) in the Gulf of Mexico (Rabalais et al. 2002).

**TABLE 1. Wetland types of the Midwest (USA) and their relative contribution for delivering ecosystem services related to water quality maintenance, habitat, hydrology, and carbon sequestration.**

<table>
<thead>
<tr>
<th>Wetland type</th>
<th>Water quality maintenance</th>
<th>Habitat</th>
<th>Hydrology</th>
<th>Carbon sequestration</th>
</tr>
</thead>
<tbody>
<tr>
<td>Riparian forest</td>
<td>high</td>
<td>high†</td>
<td>low–medium†</td>
<td>low</td>
</tr>
<tr>
<td>Floodplain forest</td>
<td>high</td>
<td>high†</td>
<td>high†</td>
<td>low</td>
</tr>
<tr>
<td>Depression</td>
<td>low</td>
<td>high§</td>
<td>medium#</td>
<td>low</td>
</tr>
<tr>
<td>Vernal pool</td>
<td>low</td>
<td>high§</td>
<td>low</td>
<td>low</td>
</tr>
<tr>
<td>Wet meadow</td>
<td>low–medium</td>
<td>medium‡</td>
<td>low–medium</td>
<td>low–medium</td>
</tr>
<tr>
<td>Fen</td>
<td>low–medium</td>
<td>high‡</td>
<td>low</td>
<td>medium</td>
</tr>
<tr>
<td>Seep</td>
<td>low</td>
<td>medium</td>
<td>low</td>
<td>low</td>
</tr>
<tr>
<td>Bog</td>
<td>low</td>
<td>high‡</td>
<td>medium#</td>
<td>high</td>
</tr>
</tbody>
</table>

† High productivity and connectivity.
‡ Flood de-synchronization.
§ Breeding grounds for herpetofauna.
¶ High plant diversity.
# Groundwater recharge.

Died nutrients, and other chemicals that they receive. This service can vary both temporally and spatially, but wetlands have consistently been shown to improve the water quality of nonpoint source runoff, and wetland losses have contributed to reductions in water quality in the region. The support of diversity

One consequence of widespread wetland losses is the decline of wetland-dependent species such as amphibians, invertebrates, and waterfowl, as well as species that are primarily terrestrial but use wetlands for refugia and subsidy (Hansson et al. 2005). In the case of amphibians, habitat loss and fragmentation are recognized as major causes of diversity declines (Hecnar and M‘Closkey 1996). Amphibians are particularly at risk due to their relatively poor dispersal abilities (Findlay and Houllahan 1997, Lehtinen et al. 1999). Two particular species of anurans that occur in the GIP and are reportedly in decline are the northern leopard frog (*Rana pipens*) and the cricket frog (*Acris crepitans blanchardi*) (Kolozsvary and Swihart 1999).

Reptiles are relatively unstudied, but data indicate that taxa such as freshwater turtles have declined as agriculture and road density have increased (Rizkalla and Swihart 2006), with predictions that wetland-dependent herpetofauna face a greater risk of extinction than other vertebrate species (White et al. 1997). Wetland invertebrates provide an important connection between primary productivity and the abundance of migratory and resident vertebrate populations, serving as important food supplies for a wide array of bird species such as bitterns, egrets, herons, shorebirds, waterfowl, and songbirds (Hershey et al. 1999). Despite this, there are few data on the status of invertebrate communities in light of wetland losses. Bird diversity is known to decline with wetland area, so wetland loss has consequences for the persistence of avian species in the region (Hershey et al. 1999). Thus, the loss of wetlands in the GIP has contributed directly to declines in regional biodiversity (Findlay and Houllahan 1997, Hershey et al. 1999).

**Flood abatement**

Wetlands naturally trap and store water, and floodplains are particularly important in water storage and flood conveyance. As floodplains and wetlands have been altered or lost, their ability to store floodwaters in the GIP has diminished. Nationally, the estimates of damage due to flooding have risen steadily since 1900, increasing from an annual mean of US$1.4 billion over the 30-year period 1903–1933 to $3.4 billion during 1963–1993 (Hey and Philippi 1995). This occurred as wetlands were drained and floodplains developed (Hey and Philippi 1995, Zedler and Kercher 2005). The Mississippi River flood of 1993 alone cost an estimated $12–$16 billion.

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Despite the extent of agricultural land use, the species richness of remaining wetlands can be high. For example, Mensing et al. (1998), in a study of 15 riparian wetlands, documented 175 plant species and 151 animal species (including birds, fish, amphibians, and macro-invertebrates), demonstrating the value of wetlands in biodiversity support.

Carbon sequestration

Hydrologic conditions that lead to the accumulation of soil carbon are a defining feature of wetlands (Bridgham et al. 2007). Carbon storage capacity varies depending on wetland type, with peatlands (freshwater wetlands with surface soil organic deposits greater than 40 cm thick, commonly called bogs and fens) storing considerably more C than wetlands with mineral soils (those with less soil organic matter). Globally, peatlands contain between 16% and 33% of the total soil carbon pool (Bridgham et al. 2001). While the majority of peatlands occur north of 50° N, there are substantial areas of peatlands in the upper GIP (Gorham 1991, Bridgham et al. 2007). Bogs receive nearly all of their water from precipitation (Fig. 2), the peat is acidic, nutrient poor, and only slightly decomposed (Bridgham et al. 2001). Fens are typically phosphorus limited but, because of groundwater inputs, they are not as acidic as bogs and contain more highly decomposed peat (Bridgham et al. 2001). The slower rate of decomposition (i.e., carbon mineralization) in bogs compared to fens (Bridgham et al. 1998) leads to deeper peat and organic C accumulation (Craft et al. 2008).

Regionally specific estimates of carbon storage in the GIP are lacking, but wetland drainage has reduced carbon stores substantially, perhaps by as much as 15 million Mg of carbon per year in North America (Bridgham et al. 2006). The loss of C storage has diminished wetlands’ role as climate regulators.

Effects of NRCS Conservation Programs and Practices on Wetland Ecosystem Services

Of the ecosystem services wetlands provide, four are particularly significant in the GIP: water quality improvement, flood abatement, biodiversity support, and carbon processing and storage. We focus here on water quality improvement, and to a lesser extent on flood abatement, biodiversity support, and carbon storage. This follows, in large part, what we know based on the available literature.

In order to restore wetland and riparian-zone acreage to the landscape, a suite of conservation practices has been developed by the USDA Natural Resources Conservation Service (NRCS) specifically to establish or manage wetlands on agricultural and associated lands. These practices are designated as: wetland creation (establishing wetlands on non-hydric soils), enhancement (typically changing the hydroperiod of an existing wetland using water control structures), restoration (establishment of wetland on hydric soils), and floodplain forests are important regulators of the flow of materials across the landscape, reducing wildlife habitat management (a system of practices that establishes or restores wildlife habitat, and may be implemented with wetland restoration, creation, or enhancement practices), and riparian forest buffer establishment (establishment of a riparian forest buffer along streams where former riparian wetlands existed or where they currently exist but in a degraded state). These practices are collectively referred to as “wetland conservation practices.” Implementation of these practices is supported by financial and technical assistance as codified in the conservation title of what has become known as the “Farm Bill” to achieve environmental or biological goals on agricultural and associated lands. In this case, the Farm Bill refers to a series of legislative acts, including the Food Security Act of 1985; the Food, Agriculture, Conservation and Trade Act of 1990; the Federal Agricultural Improvement and Reform Act of 1996; the Farm Security and Rural Investment Act of 2002; and the Food, Conservation and Energy Act of 2008. Collectively, these conservation practices have added to or enhanced more wetland acreage in the agricultural landscapes of the GIP than in any other region included in this study. The implementation of wetland conservation practices supported by a variety of Farm Bill conservation programs (e.g., Wetlands Reserve Program [WRP], Conservation Reserve Enhancement Program [CREP], Conservation Reserve Program [CRP], Wildlife Habitat Incentives Program [WHIP]) are expected to reestablish ecosystem services, providing regional benefits in terms of water quality, flood abatement, biodiversity support, and carbon storage.

In total, >1 million wetland conservation projects were implemented in the GIP, accounting for 33.4% of all practices implemented nationally between 2000 and 2006, and amounting to ~110 000 ha (1100 km2) of land. In terms of area, the wetland practices implemented are dominated by wetland restoration, followed by the establishment of wetland wildlife habitat, and riparian buffers (Fig. 3). Over 80% of wetland conservation practices in the GIP were implemented between 2004 and 2006, demonstrating the potential for landscape change over a relatively short time period (Fig. 3).

At the regional scale, we examined the distribution of conservation wetlands and riparian zones by mapping the top 100 subwatersheds in the GIP based on the total area of wetland conservation practices established in each (Fig. 4). The largest concentration of wetland conservation practices is located in the southwestern part of the GIP that largely coincides with the drainage basin of the Mississippi River. Subwatersheds that received the largest acreage of wetland conservation include the Rock River in Illinois and the Wabash River in southern Illinois and Indiana.

Water quality maintenance

Because of their high connectivity to uplands, riparian and floodplain forests are important regulators of the flow of materials across the landscape, reducing
sediment and phosphorus (P) and nitrogen (N) loads to streams and rivers (Risser 1993, Fennessy and Cronk 1997). Riparian and floodplain forests can remove substantial amounts of sediment (Table 2); for example, riparian wetlands trap up to 50 Mg·ha⁻¹·yr⁻¹, substantially more than floodplain forest’s 2–5 Mg·ha⁻¹·yr⁻¹. Sediment accumulation is low (0.2–2 Mg·ha⁻¹·yr⁻¹) in groundwater and precipitation-driven wetlands such as depressions, bogs, and fens. Phosphorus removal also is greater in riparian forests and floodplain wetlands (Table 2). And N accumulation in soils is greater in floodplains (30–300 kg N·ha⁻¹·yr⁻¹) than in bogs (80 kg N·ha⁻¹·yr⁻¹), fens–cedar swamps (30–60 kg N·ha⁻¹·yr⁻¹), or depressions (30 kg N·ha⁻¹·yr⁻¹). Constructed wetlands in the region remove relatively large amounts of sediment and nutrients, comparable to riparian and floodplain wetlands (Table 2).

Denitrification is the primary process by which nitrate is transformed by wetlands, thereby removing a key waterborne pollutant (Table 3). Because of their connectivity to lotic ecosystems, high C availability, and inflows of nitrate, denitrification is greater in riparian and floodplain wetlands than in depressions, bogs, or fens in the region (Table 3). Riparian wetlands intercept nitrate in shallow groundwater (Gilliam 1994, Hill 1996), whereas river flooding enhances denitrification of surface water nitrate in floodplain wetlands (Hernandez and Mitsch 2006). Although depressions, bogs, and fens are not as important for water quality improvement as riparian and floodplain wetlands, they are significant nutrient sinks at local scales (Craft and Casey 2000, Whigham and Jordan 2003).

In the GIP, riparian buffers and constructed wetlands are increasingly employed as the best management practices to filter sediment and nutrients and improve water quality. From a landscape perspective, studies suggest that nutrient export from stream catchments is correlated with the presence of wetlands and with the extent and characteristics of the riparian zone (e.g., buffer width, vegetation type). In Wisconsin, catchments with more wetland area (7–10% vs. <4%) had lower P yields during extreme precipitation events (Reed and Carpenter 2002). Variability in P yield was negatively correlated with riparian characteristics of width, continuity, and sinuosity, suggesting that preferential transport of nutrients to stream waters occurs in gaps in riparian corridors, such as those created by roads (Reed and Carpenter 2002). The placement of buffers within the landscape to intercept nutrients and other pollutants also is important. In Iowa, researchers calculated that 56% of riparian cells (using a 30-m grid) surveyed would receive runoff from <0.4 ha (Tomer et al. 2003), suggesting that it is important to strategically place riparian buffers in areas that will maximize interception of water, sediment, and nutrient flows.

In the agricultural Midwest, nitrate leaching is a significant problem, enriching streams and rivers (Keeney and DeLuca 1993) and contributing to hypoxia in the Gulf of Mexico (Turner and Rabalais 1991, Rabalais et al. 2002). Natural and restored wetlands, riparian areas, and constructed wetlands have the potential to reduce nitrate loadings and help alleviate this problem. Whitmire and Hamilton (2005) reported that a variety of natural wetlands in Michigan, ranging from groundwater fens to precipitation-driven bogs, have the potential to remove significant amounts of nitrate. In Illinois, grass and forested riparian buffers reduced local nitrate loadings by up to 90% (Osborne and Kovacic 1993), and forest vegetation, because its roots penetrate deeper and deliver more carbon to denitrifying bacteria, was more effective than grass for intercepting nitrate as it is preferentially transported in subsurface flow (Fennessy and Cronk 1997).

Tile drainage complicates the removal of N from agricultural runoff since wetlands typically have little opportunity to intercept N in this water (Crumpton et al. 2006). Where wetlands can intercept tile drainage, N-removal efficiencies are high. In a study of the Iowa CREP program, Crumpton et al. (2006) report that wetlands restored to intercept tile drainage removed between 25% and 78% of NO₃⁻ amounting to 368–2310 kg N·ha⁻¹·yr⁻¹. In another study in Illinois, a constructed wetland removed an average of 33% of the N in tile drainage (Miller et al. 2002). However, there was no significant removal of P or nine common herbicides (e.g., atrazine, alachlor) used in the Midwest. Phosphorus removal by constructed wetlands receiving tile drainage typically is low, as little as 2% (Kovacic et al. 2000) because much of the P (>50%) is transported bound to sediment in surface waters (Royer et al. 2006). Little is known about the relative effectiveness of different conservation practices (e.g., constructed wetland vs. wetland restoration) in improving water quality, and what the associated trade-offs might be between water quality improvement and other ecosystem services.

Constructed wetlands also have been used to remove instream pollutants. These wetlands typically receive higher loads of sediment and nutrients and, so, remove greater quantities of pollutants than natural wetlands in the region (Table 2). Four wetlands were constructed in the floodplain of the Des Plaines River (Illinois) in the late 1980s to filter sediment, nutrients, and agricultural chemicals from the river water (Kadlec and Hey 1994). Over a three-year period, the wetlands removed between 40% and 95% of incoming nitrate-N, 38–100% of the incoming sediment, 27–100% of the phosphorus, and ~50% of the herbicide atrazine (Fennessy et al. 1994, Kadlec and Hey 1994, Phipps and Crumpton 1994). Removal rates were greater in the summer than winter, illustrating how removal efficiencies vary with season. Flow rates also regulated nutrient retention as the efficiency of P removal was greater under low-flow (64–92% removal) than high-flow conditions (53–90% removal; Mitsch et al. 1995). On average, the constructed wetlands removed ~5–30 kg P·ha⁻¹·yr⁻¹, comparable to highly loaded natural
wetlands. The percentage of nitrate removal was also greater under low- than high-loading conditions in these wetlands (Phipps and Crumpton 1994).

On the Olentangy River (Ohio), constructed wetlands also remove substantial quantities of sediments and nutrients. Over a 10-year period (1994–2004), the wetlands stored an average of 43–47 Mg sediment ha\(^{-1}\) yr\(^{-1}\), 162–166 kg N ha\(^{-1}\) yr\(^{-1}\), and 33–35 kg P ha\(^{-1}\) yr\(^{-1}\) in soil (Table 2). During the same period, the wetlands removed an average of 410–470 kg N ha\(^{-1}\) yr\(^{-1}\) of nitrate-N via denitrification, with only a small amount, <1%, evolved as N\(_2\)O (Hernandez and Mitsch 2006).

![Map of wetland conservation practices](image1)

**Fig. 3.** The area of wetland conservation practices implemented in the Glaciated Interior Plains between 2000 and 2006.

![Map of top 100 watersheds](image2)

**Fig. 4.** The top 100 watersheds in the Glaciated Interior Plains (by hectares and acres) of wetland conservation practices implemented through Farm Bill conservation programs. The conservation practices include Wetland Creation, Wetland Restoration, Wetland Enhancement, Riparian Forest Buffer, and Wetland Wildlife Habitat Management.
Not all constructed wetlands are as effective for pollutant removal. Sidle et al. (2000) reported that wetlands constructed in the Kankakee River (Indiana) were less effective for nitrate removal than natural wetlands because of nonuniform subsurface flow and short hydrologic residence time. For constructed and restored surface water wetlands to be effective, they must have high water storage capacity to retain nitrate during high-discharge events, when greater than 50% of the nitrate is exported from adjacent uplands (Royer et al. 2006).

**Flood abatement**

Wetlands are known to regulate the movement of water through watersheds via a combination of processes such as water storage, evapotranspiration, and gradual release that augments stream base flow in dry seasons (Brauman et al. 2007). Increasing wetland area in a watershed will almost always decrease its surface water discharge due to the effects of vegetation alone, particularly forests (Brooks et al. 2006, Brauman et al. 2007). For example, Novitski (1985), working in the Northeastern United States, found peak flows were an average of 50% lower in watersheds with 4% or greater wetland area compared to those with less wetland coverage. Models of the relationships between the flood storage capacity of wetlands as a percentage of total land area in a watershed and flood peak reduction have shown a threshold of 10%, such that, in watersheds with <10% wetland area, even small additional losses in

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**Table 2. Rates of sediment, P, N, and C accumulation in soils of Midwestern wetlands.**

<table>
<thead>
<tr>
<th>Wetland type and state</th>
<th>Sediment (Mg ha⁻¹ yr⁻¹)</th>
<th>P (kg ha⁻¹ yr⁻¹)</th>
<th>N (kg ha⁻¹ yr⁻¹)</th>
<th>Organic C (kg ha⁻¹ yr⁻¹)</th>
<th>Source</th>
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<tr>
<td>Riparian</td>
<td></td>
<td></td>
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<tr>
<td>Wisconsin</td>
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<td>82</td>
<td>524</td>
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<td>50</td>
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<td></td>
<td>Heimann and Roell (2000)</td>
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<td>Floodplain</td>
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<td></td>
</tr>
<tr>
<td>Illinois</td>
<td>34</td>
<td></td>
<td></td>
<td></td>
<td>Mitsh et al. (1979)</td>
</tr>
<tr>
<td>Wisconsin</td>
<td>5</td>
<td>11</td>
<td>27</td>
<td>540†</td>
<td>Johnston et al. (1984)</td>
</tr>
<tr>
<td>Wet meadow</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Marsh</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Indiana depressional</td>
<td>1.8</td>
<td>30†</td>
<td>610</td>
<td></td>
<td>C data from Craft et al. (2008); sediment, P, and N data from C. B. Craft (unpublished data)</td>
</tr>
<tr>
<td>Fen</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Indiana</td>
<td>0.9</td>
<td>2</td>
<td>61</td>
<td>900</td>
<td>N and P data from Craft and Schubauer-Bergin (2006), C data from C. B. Craft (unpublished data)</td>
</tr>
<tr>
<td>Michigan</td>
<td>0.2</td>
<td>1–9†</td>
<td>30</td>
<td>420</td>
<td>Graham et al. (2005), Richardson and Marshall (1986)</td>
</tr>
<tr>
<td>Cedar swamp</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Michigan</td>
<td>0.3</td>
<td>36</td>
<td>950</td>
<td></td>
<td>C data from Craft et al. (2008); sediment, P, and N data from C. B. Craft (unpublished data)</td>
</tr>
<tr>
<td>Bog</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Minnesota</td>
<td>0.5</td>
<td>7–12</td>
<td>790–1980</td>
<td></td>
<td>Wieder et al. (1994), Urban and Eisenreich (1988)</td>
</tr>
<tr>
<td>Michigan</td>
<td>0.3</td>
<td>80</td>
<td>1320</td>
<td></td>
<td>C data from Craft et al. (2008); sediment, P, and N data from C. B. Craft (unpublished data)</td>
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<tr>
<td>Constructed marsh</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Note: Numbers in boldface represent accumulation in highly loaded wetlands.

† Calculated assuming C:N = 20.

‡ Richardson and Marshall (1986) found 9 kg ha⁻¹ yr⁻¹ from a fen receiving wastewater.
area can have major effects on flood flows (Johnston et al. 1990). If wetlands make up 10% or more of a watershed, flooding is reduced and stream base flows are better maintained; for instance, Hey and Philippi (1995) calculated that restoring 5.3 million ha of wetlands in the Upper Mississippi River watershed (10% of the land area of the watershed) could have stored enough water to accommodate the 1993 Mississippi floods, thereby substantially reducing the $16 billion in flood damages that resulted. Beyond this, there are few direct estimates of the ability of wetlands to reduce flooding in the GIP, nor are there spatially explicit data available for conservation wetlands that would allow calculation of the contribution of these sites in reducing runoff and downstream flooding.

Research to increase our understanding of the aggregate effects of wetlands on flood flows would also help define the most strategic placement of wetlands in a watershed to maximize these services. Based on acreage alone, the wetlands established through conservation practices and programs to date may have relatively small effects at the large scale, but almost certainly have beneficial effects at the local scale, particularly in watersheds where the area of wetlands established is relatively high (Fig. 4). For instance, wetlands located in headwater areas within a watershed can effectively abate flooding depending on the ratio of their storage relative to the volume of floodwater (Potter 1994). These landscape approaches to wetland services allow for the setting of provisional targets for the restoration or enhancement of wetland acreage in flood-prone areas.

Biodiversity

While there have been few systematic data collected on the establishment of animal species as a result of wetland conservation practices, the available literature indicates that the establishment of wetland-dependent species is rapid across many taxonomic groups including invertebrates, herpetofauna, fish, and birds (Rewa 2005). In some cases, species richness has been shown to approach natural levels; for example, comparisons of natural with newly restored wetlands has shown that amphibian, avian, and invertebrate species richness was similar, or in some cases, higher in the created sites (e.g., Balcombe et al. 2005a, b), although this is sometimes due to a predominance of nonnative or tolerant species. With respect to these taxonomic groups (including plants, which are not discussed here), there are no data that allow us to fully describe the benefits of the various wetland conservation practices, or to distinguish how they might differ in habitat value.

Despite the importance of macro-invertebrates in wetlands, there are few studies documenting their successional patterns or community development. Invertebrates are diverse both in terms of species richness and the trophic levels they occupy (herbivores, detritivores, predators/carnivores), and so they are involved in multiple ecosystem processes. Several studies have found invertebrate species richness to be similar in natural and restored wetlands (e.g., Balcombe et al. 2005b). However, a study comparing macro-invertebrate community structure in restored and natural wetlands in Ohio found significantly higher total taxa richness, particularly for chironomids, in the natural sites, with a higher relative abundance of dipterans and tolerant snails in the restored sites (Fennessy et al. 2004). In Wisconsin, Dodson and Lillie (2001) found that zooplankton taxa richness was similar in natural (with an average of 7.3 taxa per site) and restored wetlands (7.2 taxa). In this study, taxa richness was positively correlated with time since restoration.

Bird species response to wetland restoration is relatively well documented. Bird species are shown to rapidly colonize sites, and while reports vary, many projects report that avian species richness levels approach that of natural wetlands, including those of management concern in Iowa and migratory species in Ohio (Mitsch et al. 1998, Fletcher and Koford 2003). A constructed wetland located on floodplain of the Olentangy River in Ohio provided habitat for a total of 126 bird species (wetland and adjacent terrestrial habitat) by the fifth year following construction (Mitsch et al. 1998).

The capacity of wetlands to support amphibian populations depends on both the conditions within the site and the characteristics of the surrounding landscape, including the presence of terrestrial forested buffers, distance to nearest neighbor wetlands, road density, and presence of corridors connecting habitat areas (Knutson et al. 1999, Lehtinen et al. 1999, Weyrauch and Grubb 2004). Both road density and distance to the nearest neighbor wetlands in agricultural landscapes were
significant predictors of amphibian species richness in the GIP (Lehtinen et al. 1999).

Microhabitats and trophic interactions affect species establishment. In a study of Ohio wetlands, Porej et al. (2004) found that amphibian diversity was significantly higher in restored wetlands with shallow water and without predaceous fish. Vegetated shallows are important for floral and faunal diversity and are nearly always present in natural wetlands. The overall structure of the amphibian community was also different in the restored sites, with a dominance of some species, e.g., bullfrog (*Rana catesbeiana*), green frog (*Rana clamitans*), and toads, and the near absence of other species, such as spring peepers (*Pseudacris crucifer*), western chorus frogs (*Pseudacris triseriata*), and most salamanders. Ultimately, wetland practices that provide sites with seasonal inundation and an upland buffer will provide habitat for herpetofauna. With this, as with other taxonomic groups, details on the relationships between wetland conservation practices, landscape characteristics, and the provision of habitat are yet to be determined.

**Carbon sequestration**

Soils are a major reservoir of organic matter and thus an important sink of carbon. Compared to agricultural soils, which contain an average of 0.5–2% C with up to 5% C, wetland soils can accumulate up to 30–40% C (Lal et al. 1995). Wetlands generally sequester carbon at higher rates than terrestrial soils in the region, and peat-accumulating wetlands, bogs, and to a lesser extent fens, have the greatest capacity to accumulate and store carbon (Table 2).

The importance of Midwestern wetlands for carbon sequestration has been understudied relative to terrestrial ecosystems, where vast areas potentially are available for restoration. However, in the conterminous United States, C sequestration in wetlands with mineral soils is estimated to be 5.3 Pg (1 Pg = 10¹² g = 1 billion metric tons), with 6.6 Pg stored in non-permafrost peatlands, a small proportion of the 529 Pg of C stored in wetland soils globally (Bridgham et al. 2006). Floodplain wetlands in the region sequester C at rates comparable to bogs and fens, whereas depressional wetlands have lower rates of C sequestration (Table 2).

Although wetlands sequester more carbon than terrestrial ecosystems per unit area, they occupy only a fraction of the Midwest landscape, and C sequestration in soil is a slow process relative to biomass accrual. For this reason, wetlands are not considered an important short-term restoration strategy for sequestering carbon. There is also a trade-off between the capacity of wetlands to absorb carbon dioxide and the fact that they are a source of methane, a potent greenhouse gas. Estimates vary, but it is possible that any gains in C sequestration due to wetland restoration could be offset by methane emissions (Bridgham et al. 2006). Further complications arise from the fact that nutrient enrichment stimulates peat accretion and C accumulation in wetland soils of the GIP and elsewhere (Craft and Richardson 1993). For example, in Indiana, a natural floodplain marsh receiving treated wastewater sequestered three times more C (3700 kg ha⁻¹ yr⁻¹) than a comparable unenriched floodplain marsh (1200 kg ha⁻¹ yr⁻¹) (Table 2). The effect of nutrient enrichment on CH₄ emissions is in the GIP is largely unknown.

The pattern of carbon fluxes over the long term is important to the question of whether restored wetlands will be a carbon sink. Bridgham et al. (2006) estimate that the historical destruction of wetlands (through drainage, et cetera) had the largest impact of carbon fluxes, moving C from soil to the atmosphere. Until more data are available, where wetlands are created or restored, their potential to sequester C should not be overlooked (Table 2).

**Contributions of Conservation Practices to Water Quality in the GIP**

Wetlands serve an important role in nutrient management at the landscape scale. Restoring wetlands to improve water quality requires a landscape approach that maximizes the ability of sites to capture and process diffuse runoff. At the site-scale riparian zones, even narrow bands (e.g., <10 m) adjacent to streams, ditches, or rivers can remove up to 90% of N (Zedler and Kercher 2005). Although relatively little is known about the links between the placement of wetlands within a watershed and the accumulation of services at the watershed scale, the extent of wetland practices implemented can be used to make an initial characterization of the ecosystem services provided.

The degradation of water quality in the Upper Mississippi has been extensively documented. In an investigation of N loading to the Mississippi River, Goolsby et al. (1999) delineated 42 subwatersheds to provide detailed information on N export and identify those with abnormally high outflows. Nitrogen was the focus as it has been identified most responsible for hypoxia in the Gulf of Mexico (Rabalais et al. 2002). Sixteen of these watersheds lie wholly or partially in the GIP (Fig. 5), including some with the highest nutrient export rates in the entire basin. For example, the upper Illinois River exports more N per unit area than any other subwatershed in the Mississippi Basin, yielding 3120 kg N km⁻² yr⁻¹. This is followed by the Iowa and Skunk Rivers at 2750 and 2290 kg N km⁻² yr⁻¹, respectively.

We used estimates of the nutrient retention rates for the various wetland conservation practices (Table 2), along with an estimate of the area of wetland conservation practices implemented within the portion of the GIP that lies in the Mississippi River watershed, to estimate their cumulative nutrient retention ability (Table 4). This provides an estimate of the cumulative nutrient retention capacity of conservation wetlands put in place through Farm Bill conservation programs (e.g.,
WRP, CREP, CRP, Technical Assistance), and their overall ability to reduce sediment, N, and P export to the Gulf of Mexico. Nutrient retention rates vary with the type and acreage of wetlands involved. Reflecting their ability to process large amounts of N, riparian buffer strips remove more N than any other practice, totaling an estimated 6.5 million kg/yr. Wetland restoration accounts for another 4.5 million kg/yr. In total, conservation practices are estimated to retain nearly 14.6 million kg/yr. The total annual load to the Gulf of Mexico ranges between 1.5 and 1.6 million Mg of N (Goolsby et al. 1999, Alexander et al. 2008); therefore, the amount of N intercepted by conservation wetlands amounts to an estimated 1.0% of the total N reaching the Gulf of Mexico annually. Given that most of the wetland area in the GIP was created over a three-year period (2004–2006), these estimates show the potential for the restoration of ecosystem services that are measurable at the watershed scale.

Sediment and P retention are also substantial. Wetland restoration projects account for the largest sink for sediments: Collectively, these sites retain an estimated 1.5 million Mg of sediment annually (Table 4), representing an estimated 1.8% of the total sediment load that flows from the Mississippi River each year. Phosphorus retention is estimated to account for 0.8% of the total annual load in the Mississippi River.

The water quality benefits of nutrient interception can also be made using estimates of the relative proportion of nutrients that are generated within the GIP region itself, calculated based on the proportion (15.8%) of the GIP that lies within the Mississippi River watershed. In this case, the estimated nutrient retention capacity of conservation wetlands amounts to 6.8% of N, 4.9% of P, and 11.5% of the sediment exported from the region annually. These estimates demonstrate the potential for reestablishing ecosystem services through conservation practices based on wetland and riparian-zone establishment and management.

In response to hypoxia in the Gulf, nutrient management strategies were set forth in the “Action plan for reducing, mitigating, and controlling hypoxia in the Northern Gulf of Mexico” (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force 2001, Rabalais et al. 2002). The action plan established the goal of reducing the five-year running average of the areal extent of the hypoxic zone to 5000 km² or less by 2015. Initial estimates set the level of N load reductions needed to meet this goal at 30% below the 1980–1996 average (Rabalais et al. 2002, Scaivia et al. 2003). Subsequent models developed during a reassessment of the action plan, required every five years, have shown that, while a 30% reduction in N loads would lead to a 20–60% reduction in the extent of hypoxia, it will require a 40–
45% nitrogen load reduction to meet the 5000 km² goal (Scavia et al. 2003, Justic et al. 2007, Scavia and Donnelly 2007). Of the 31 states that contribute to the Mississippi River flow, nine states (Illinois, Iowa, Indiana, Missouri, Arkansas, Kentucky, Tennessee, Ohio, and Mississippi) deliver 75% of the total N and P to the Gulf of Mexico (Alexander et al. 2008). The GIP contains all or parts of five of these nine states (Illinois, Iowa, Indiana, Missouri, and Ohio). A focused program of wetland restoration in the GIP region has the potential to reduce seasonal hypoxia by helping to meet targets for reductions in nutrient exports.

One component of a watershed-based approach to accomplish the goal of protecting downstream water quality is to promote large-scale efforts to create and restore wetlands (Table 5). Mitsch et al. (2001) estimated that the creation and restoration of 2.1–5.3 million ha of wetlands in the Mississippi River Basin (0.7–1.8% of the basin) could reduce nitrate loadings to the Gulf of Mexico by 300–800 × 10^3 Mg/yr, or by 18–50%, based on annual loads of 1.5–1.6 million Mg/yr. Likewise, Hey and Philipp (1995) recommended that 5.3 million ha of restored wetlands would hold the floodwater equal to the 1993 Mississippi River flood. An earlier study called for the restoration or creation of 10 million ha of wetlands and riparian zones (3.4% of the basin), concluding that this would reduce nitrogen in the river by an estimated 40% (Mitsch et al. 1999). Extrapolating from our broad estimates of the N removed by NRCS conservation practices (Table 4), 1.3 million ha (~3.1 million acres) would be required to achieve a 40% reduction in N. While these numbers will undoubtedly be refined, these studies place the need for restoration at between 3 and 10 million ha, or 1.0–3.4% of the land area of the basin. If reductions in N loads of 40% could be achieved, the analysis conducted as a result of the

<table>
<thead>
<tr>
<th>Practice</th>
<th>Total area implemented (ha)</th>
<th>Sediment retained (Mg/yr)†</th>
<th>Nitrogen retained (kg/yr)‡</th>
<th>Phosphorus retained (kg/yr)§</th>
<th>Carbon retained (kg/yr)¶</th>
</tr>
</thead>
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<tr>
<td>Constructed Wetland</td>
<td>22</td>
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<td>50.341</td>
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<td>4528.523</td>
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<td>15.095.076</td>
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<td>302.837</td>
<td>3028.373</td>
<td>302.837</td>
<td>15.141.867</td>
</tr>
<tr>
<td>Total</td>
<td>77652</td>
<td>2550.077</td>
<td>14574.247</td>
<td>1088.710</td>
<td>38826.157</td>
</tr>
</tbody>
</table>

Percentage of total nutrient load in Mississippi River retained: NA 1.8# 1.0|| 0.8† 0.8‡ 
Relative percentage of nutrient load from GIP region retained: NA 11.5 6.8 4.9 38.826 157

Note: Conservation practice area data were supplied by NRCS. NA indicates not applicable; ellipses indicate that data were not available.
† Assumes mean retention rate of 50 Mg ha⁻¹ yr⁻¹; 10 Mg ha⁻¹ yr⁻¹ for enhancement and wildlife habitat activities (see Table 2).
‡ Assumes mean retention of 500 kg ha⁻¹ yr⁻¹ for riparian buffers; 150 kg ha⁻¹ yr⁻¹ for constructed, created, and restored wetlands; and 100 kg ha⁻¹ yr⁻¹ for enhancement and wildlife habitat (see Table 2).
§ Assumes mean retention rate of 10 kg ha⁻¹ yr⁻¹ for riparian buffers; 20 kg ha⁻¹ yr⁻¹ for constructed, created, and restored wetlands; and 10 kg ha⁻¹ yr⁻¹ for enhancement and wildlife habitat (see Table 2).
# Assumes mean annual sediment export of 144 million Mg to the Gulf of Mexico.
|| Assumes mean annual N export of 1.4 million Mg to the Gulf of Mexico.
†† Assumes mean annual P export of 0.14 million Mg to the Gulf of Mexico.

<table>
<thead>
<tr>
<th>Activity and location</th>
<th>Percentage of watershed converted</th>
<th>Reduction in load (%)</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Riparian restoration (USA)†</td>
<td>0.7–1.8</td>
<td>19–50 (N)</td>
<td>Mitsch et al. (2001)</td>
</tr>
<tr>
<td>Wetland creation and restoration (USA)†</td>
<td>2.7–6.6</td>
<td>19–50 (N)</td>
<td>Mitsch et al. (2001)</td>
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<tr>
<td>Wetland creation and restoration (USA)‡</td>
<td>5</td>
<td>46 (N)</td>
<td>Kovacic et al. (2006)</td>
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<tr>
<td>Wetland creation and restoration (Sweden)‡</td>
<td>5–10</td>
<td>25–50 (N)</td>
<td>Tonderski et al. (2005)</td>
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<tr>
<td>Wetlands (China)†</td>
<td>5</td>
<td>&gt;90 (P)</td>
<td>Verhoeven et al. (2006)</td>
</tr>
</tbody>
</table>

Note: Abbreviations are: N, nitrogen; P, phosphorus.
† All activities in the United States were carried out in the Mississippi River Basin.
‡ Involves conversion of arable land to wetland.
action plan indicates that this could help meet Gulf of Mexico water quality goals.

The spatial distribution of conservation projects will be a key factor in successfully reducing nutrient loads in surface waters. By targeting restoration in certain high-nutrient watersheds like the Illinois River Basin, even greater reductions could be achieved with less effort (Mitsch et al. 2001, Kovacic et al. 2006). While studies to quantify the effects of the spatial distribution of conservation practices within a watershed are underway in projects such as the NRCS “benchmark” watershed studies (King et al. 2008), results to date are limited and questions remain on the watershed-scale benefits of project implementation.

One crucial question regarding ecosystem services is whether the implementation of conservation practices coincides with areas that are in most need of restoration, such as locating conservation wetlands appropriately to treat high-nutrient runoff. Using the subwatersheds to evaluate nutrient export to the Mississippi River (Goolsby et al. 1999), we found a strong correlation \( r = 0.81 \) between the total area of NRCS conservation practices implemented between 2000 and 2006 and the estimated nitrogen export from each subwatershed prior to the implementation of these projects (Fig. 6), suggesting that, at least for water quality improvement, conservation practices are successfully targeting watersheds that are exporting the most N. At its best, the provision of ecosystem services through implementation of wetland conservation practices will result in improved environmental quality and human health in terms of improved downstream water quality. What is lacking in the GIP is specific data to demonstrate that these conservation practices are measurably decreasing N export at the subwatershed scale, and whether or not wetlands are being sited within watersheds in ways that maximize water quality improvement (White and Fennessy 2005).

**Conclusions**

More than 80% of the acreage involving Wetland Restoration, Creation, Enhancement, the establishment of Riparian Forest Buffers, and Wetland Wildlife Habitat Management conservation practices implemented in the period between 2000 and 2006 was put in place in just three years (2004–2006), demonstrating the potential for wetland practices to rapidly restore ecosystem services that are measurable at the watershed scale. However, many subwatersheds in the Glaciated Interior Plains were untouched by these practices and associated Farm Bill conservation programs; thus, they offer the potential for further conservation efforts to be implemented in the region. The preliminary results presented here indicate generally that the quantity of ecosystem services delivered regionally can be increased over remarkably short time periods. Our analysis points to the need for indicators that can document the ability of conservation practices to deliver ecosystem services and how the spatial distribution of conservation practices on the landscape affects the delivery of those services. In many cases landowners have choices about which practices might best suit their land or needs. Decision making of this sort would be strengthened if we understood the trade-offs in the relative degree of services delivered by one conservation practice (e.g., wetland restoration) over another (wetland wildlife habitat management). At this point there is no means to distinguish the benefits provided by individual practices or the trade-offs between the provisioning of different ecosystem services (e.g., water purification vs. biodiversity support). This is particularly true for carbon sequestration and potential changes in carbon dynamics due to altered nutrient loadings and issues related to climate change (e.g., CH\(_4\) emissions). Future establishment of conservation projects might take advantage of the opportunity to impose whole watershed experiments by systematically testing the effects of the quantity and spatial distribution of wetland projects (or lack thereof) on a subwatershed basis. This approach is being used in several of the NRCS benchmark studies designed to investigate the benefits of watershed-scale restoration (King et al. 2008). Focusing research on these questions will help us determine whether, and to what extent, wetland and riparian restoration projects on agricultural lands are providing critical wetland ecosystem services.

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