Wetlands at your service: reducing impacts of agriculture at the watershed scale

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In the Upper Midwestern region of the US, three ecosystem services (flood abatement, water quality improvement, and biodiversity support) declined when about 60% of the region's historical wetland area was drained, mostly for agriculture. Some of the lost services could potentially be regained through wetland restoration measures authorized in the 2002 Farm Bill. Because no single wetland can provide all ecosystem services indefinitely, ecologists can help to identify combinations of projects that will best restore ecosystem services within watersheds. "Strategic" restoration would use an adaptive management approach, targeting former wetlands with marginal crop production, and prioritizing the location, size, and type of wetland needed for a watershed to provide optimal levels of all three services. Given that the Farm Bill includes over $1 billion per year to conserve natural resources on agricultural lands, we are in an excellent position to increase the effectiveness of wetland restoration.

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Farming has transformed the landscape of the US Midwest by converting upland and wetland habitats to crops and pastures, simultaneously changing hydrologic conditions and water quality, and threatening the region's biodiversity (Table 1). Where corn is the dominant crop (in Iowa, Illinois, and Indiana, in particular), extensive networks of perforated pipes and drainage ditches were installed to drain wetlands. As a result, surface and ground waters flow directly into rivers, and cultivation and fertilization release sediments, phosphorus (P), and nitrogen (N) into downstream waters (Howarth *et al.* 2002). About 43% of the nation's N fertilizer is applied to the corn fields that cover only 21% of the agricultural land, and at least 35% runs off into streams, lakes, and ultimately coastal waters (Crumpton *et al.* 1993; Howarth *et al.* 2002). Excess P causes algal blooms in inland lakes, and excess N causes seasonal hypoxia (low oxygen levels) in coastal waters. Although farming practices have improved since the 1930s Dust Bowl years, serious problems remain. Even where farmers reduce fertilizer applications, agricultural fields discharge more surface runoff and nutrients than do undisturbed soils (Woltemade 2000; Howarth *et al.* 2002). In the large Mississippi drainage basin, agricultural activities contribute up to 65% of the nutrients that reach the Mississippi River (NSTC 2000).

Aquatic and wetland species, in turn, are affected by the poor water quality caused by the plowing of habitat and the channelization of streams. In Missouri and Illinois, for example, 10–11% of fish species are endangered (Stein and Flack 1997). Low water quality can threaten the survival of rare plant communities, such as those found in wetlands; The Nature Conservancy lists several types of wetland as "rare" in 12 Midwestern states (Grossman *et al.* 1994). As a result of runoff from urban and agricultural

In a nutshell:

- When large areas of wetland are drained for agriculture, the ecosystem services these wetlands performed are lost
- Lost services include flood abatement, improved water quality, and support for biodiversity
- These services could be restored through careful planning and restoration of former wetlands at sites unprofitable for farming
- Scientists can help plan restoration by adapting existing landscape-design models to agricultural landscapes, proposing alternative strategies, and evaluating their effectiveness
- An adaptive, science-based process would increase the effectiveness of funds spent on restoration

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Table 1. Historical effects of agriculture on Upper Midwest watersheds

1. Hydrological conditions, which determine where wetlands occur as well as many of their structural and functional attributes, changed when cultivation
    - increased the amount of storm runoff
    - elevated peak flows in streams
    - exacerbated stream channel erosion
2. Water quality declined when runoff
    - increased the delivery of sediment and nutrients (notably P and N) to streams, lakes, wetlands, and estuaries
    - carried toxic materials downstream
3. Biodiversity support declined when agricultural activities
    - fragmented blocks of habitat for vegetation and wildlife
    - reduced or eradicated sensitive species
    - facilitated the establishment and expansion of invasive species
    - homogenized microtopography

Figure 1. *Monotypic stand of reed canary grass* (Phalaris arundinacea) *in the Univerisity of Wisconsin–Madison Arboretum. Researchers are looking for other species of plants that can coexist with this aggressive invader.*

lands, weedy species such as cattails (*Typha* spp), reed canary grass (*Phalaris arundinacea*), purple loosestrife (*Lythrum salicaria*), and common reed (*Phragmites australis*) spread aggressively and displace native plant species (Mensing *et al.* 1998; Galatowitsch *et al.* 1999; Werner and Zedler 2002; Woo and Zedler 2002; Figure 1).

Wetlands can abate flooding, improve water quality, and support biodiversity (Neely and Baker 1989; Crumpton *et al.* 1993; Richardson and Craft 1993; Bedford 1999; Keddy 2000). Wetlands contribute as much as 40% of the earth's renewable ecosystem services, even though they cover only 1.5% of the planet's surface (Table 2). The ability of Upper Midwestern ecosystems to absorb and lessen agricultural impacts on downstream waters has certainly been impaired by the loss of over 10 million ha of wetlands (Hey and Philippi 1995; Table 3). Evidence suggests that when less than 10–20% of a landscape's wetlands remain, the risk of flooding and eutrophication increases (Cedfeldt *et al.* 2000). In Iowa, Illinois, and Indiana, less than 5% of each state's area remains in wetlands (Table 3), and these wetlands may be functionally impaired. Of 42 Mississippi River sub-basins, the 12 that yield the most nitrate are in Illinois, Iowa, Minnesota, Wisconsin, Indiana, and Ohio (Goolsby *et al.* 1999). Today, the extent of the Gulf of Mexico's oxygen-starved "dead zone" (up to 20 000 km^2) is a barometer of lost ecosystem services upstream of the affected area (Rabalais *et al.* 1999; NSTC 2000).

Table 2. The value of wetland services (based on Costanza *et al.* 1997)

Renewable ecosystem service		*$/ha/yr*	*$billion/yr*
Hydrologic services	Water regulation	15–30	
	Water supply	3800–7600	
	Gas regulation	38–265	
Water quality services	Nutrient cycling	3677–21 100	
	Waste treatment	58–6696	
Biodiversity services	Biological control	5–78	
	Habitat/refugia	8–439	
	Food production	47–521	
	Raw materials	2–162	
	Recreation	82–3008	
	Cultural	1–1761	
	Disturbance regulation	567–7240	
Global totals	Coastal wetlands		8286
	Inland wetlands		4879
	Total for global wetlands		13 165
Total global ecosystem services for entire globe			33 268
Percentage from wetlands			**39.6%**

All shallow-water habitats (tidal marshes and mangroves, swamps and floodplains, estuaries, seagrass/algal beds, and coral reefs) are included in the calculations

Regaining ecosystem services in agricultural watersheds

The trends mentioned above can be reversed by putting wetlands back into service. Because crop yields are often marginal on former wetlands, it makes sense for farmers to offer these lands for restoration. Furthermore, because restoration can benefit society at large, it makes sense for taxpayers to compensate the farmers for restoration on their land. The 2002 US Farm Bill offers landowners the opportunity to set aside and restore former wetlands, through initiatives such as the Wetlands Reserve Program. As yet, however, the US Department of Agriculture has no clear strategy for restoring ecological services at the watershed (catchment) scale (Woltemade 2000).

Not all wetlands perform all services equally well, so we need a comprehensive restoration strategy. Large wetlands, for example, often support many bird species, especially in sites next to large areas of upland habitat (Mensing *et al.* 1998), whereas small wetland remnants may harbor rare plants. Upstream wetlands trap few nutrients, while downstream wetlands in key watershed positions can remove up to 80% of inflowing nitrates (Crumpton *et al.* 1993). We need to prioritize the types of wetlands to be restored, the total area needed, and the best locations for restoration, in order to restore ecosystem services such as biodiversity support, nutrient removal, and flood abatement at water-

Table 3. Historical changes in wetland area in five Upper Midwestern states

	IL	*IA*	*MN*	*MO*	*WI*	*Total*
1770s (thousands of ha)	3323	1619	6099	1960	3966	16 967
1980s (thousands of ha)	508	171	3521	260	2158	6618
Loss (thousands of ha)	2815	1448	2578	1700	1808	10 349
Loss (%)	85	89	42	87	46	61
Wetlands remaining in 1980 (% of state area)	3.5	1.2	16.2	1.4	14.8	

IL = Illinois, IA = Iowa, MN = Minnesota, MO = Missouri, WI = Wisconsin.
Dahl 1990

shed and river basin scales. On the other hand, if government agencies simply approve farmers' requests to set land aside on a first-come first-served basis, or if they target only large project areas in a "bigger-is-better" approach, these actions will probably compromise potential benefits.

Science can help this prioritization. Benefits have been estimated for some types of wetlands in specific settings (Potter 1994; Woltemade 2000; Shaaftsma *et al.* 2000), but it is still difficult to predict either how much of a wetland must be restored to achieve combined targets (Crumpton and Baker 1993), or by how much each ecosystem service would increase after drained fields are restored. Hey and Philippi (1995) suggested that restoring 5.3 million ha of wetlands in the upper Mississippi River Basin would significantly reduce flooding, while Mitsch *et al.* (2001) suggested that restoring 2.1–5.3 million ha of wetland and 7.8–20.0 ha of bottomland hardwood forest within the 300 million ha Mississippi River Basin would measurably reduce N flow into the Gulf of Mexico. Estimates for the conservation of wetland biodiversity are less precise. The Nature Conservancy's ecoregional conservation plans typically call for 15–25% of a landscape to be protected (Stein *et al.* 2000). For the 50 million ha basin of the upper Mississippi River, in which many watersheds retain only about 4% of their natural vegetation, a reasonable target would be to restore 1–5 million ha of wetland.

Most agricultural watersheds contain a number of former wetlands (Figure 2a). Given the various ways restoration efforts can be prioritized (Figure 2b), how might these initiatives be structured to maximize overall benefits (Figure 2c)? What size wetlands should we target, where should they be located, and what is the optimal configuration of sites within a watershed? A review of the relevant literature suggests approaches that maximize individual wetland services, although each may involve trade-offs with other functions.

Abating floods

Wetlands of various sizes and at different locations play complementary roles in moderating or preventing floods, because small wetlands high in a watershed can reduce and delay flood peaks by temporarily storing water, while large impoundments downstream can be managed to reduce peak flood levels (Mitsch 1992; Potter 1994; Hey *et al.* 2002). The ability of small, widely distributed wetlands to abate flooding depends on the amount of storage relative to the volume of floodwater, as well as the wetland's capacity for evapotranspiration (loss of water by evaporation) and infiltration (absorption of water) (Potter 1994). The restoration of prairie potholes by plugging drainage ditches, removing tiles, and excavating sediments, could therefore alleviate the effects of smaller, more frequent floods.

A substantial reduction in damages from larger, less frequent floods requires extensive downstream storage systems. Large riparian areas, drained and banked with levees for farming or other use within the 100-year flood zone, would be necessary to abate such floods. In each case, inflows and outflows need to be controlled, so that the reservoir capacity is available before the predicted peak water level, when it is most needed (Hey *et al.* 2002). If a levee is not present, Hey *et al.* (2002) suggest building 1-meter dams in selected locations in the Upper Midwest and keeping the water levels in the resulting "wetland reservoirs" low. When floodwaters rise, the excess could be allowed to flow into the wetland, where it would be held until the flood risk passed.

Wetlands used to store water for flood abatement experience rapid rises in water level as well as delays in both the peak and drying phases. The effects of these novel inundation regimes on ecosystem functions are unclear. Managers should therefore not assume that having large wetland reservoirs downstream will necessarily enhance biodiversity or lead to a substantial improvement in water quality (Figure 2b).

Improving water quality

Wetlands can and should be restored to remove contaminants, such as nutrients, from water (van der Valk and Jolly 1992). Kadlec and Knight (1996) showed clear evidence that wetlands can be used to treat urban wastewater, but this system is not readily transferable to agricultural lands because agricultural wetlands receive more pulsed flows, less organic matter, and more sediment than urban wastewater wetlands (Baker 1992; Woltemade 2000).

As runoff flows into wetlands, sediment and P can be trapped, together with settling particles. However, there is a limit to how much sediment and P can be removed, even though N can continue to be removed indefinitely, if suitable conditions are established to promote the complex sequence of microbial actions that convert nitrates to harmless N_2 gas (Schaafsma *et al.* 2000; Hey 2002). Hey (2002) advocates "nitrogen farming" – the restoration of wetlands specifically to reduce nitrate flows to the Mississippi River. Important controlling factors for such wetlands are the rate of inflow of N-rich water, the residence time of the water (how long it remains in the wet-

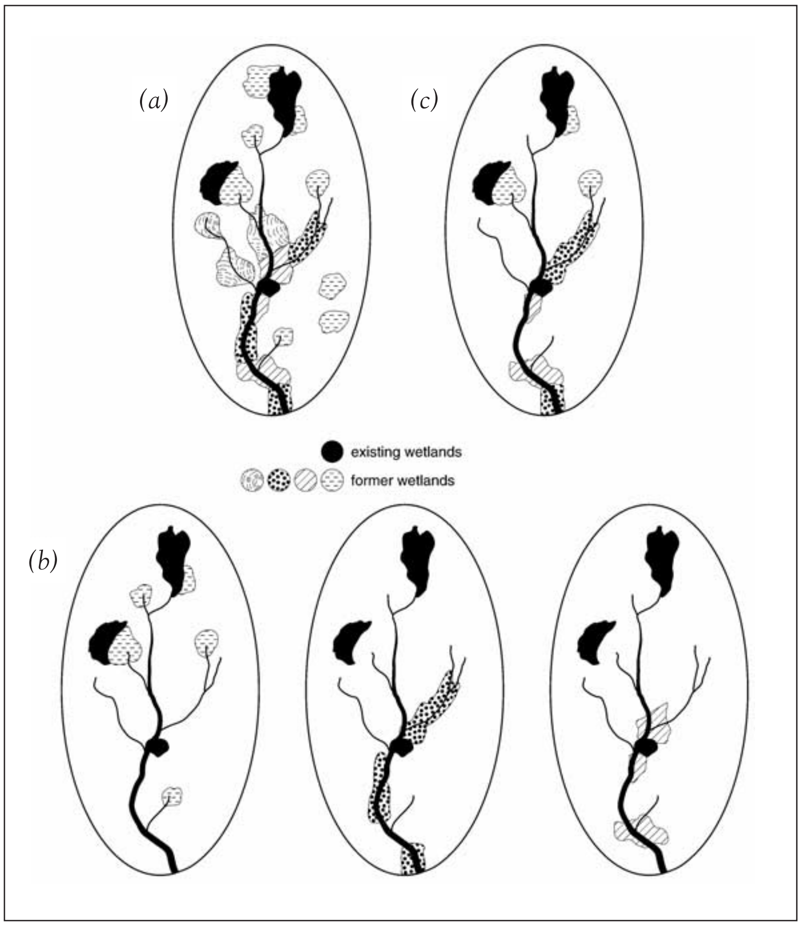


Figure 2. *Stylized graphic of an agriculturally dominated watershed. (a) Historical wetlands, with existing remnant habitat blocks indicated in black. (b) Alternative restoration scenarios designed to maximize individual ecosystem services, extrapolating from the limited literature on functions of agricultural wetlands. Left: biodiversity might benefit most from large wetlands near existing habitat blocks. Center: flood abatement may be most effective where riparian floodplains are restored. Right: water quality could be improved most by restoring wetlands downstream of tributaries with the highest nutrient-loading rates. (c) Multi-purpose design for restoring wetlands. The highest priority might go to sites identified in two or more of the designs in (b). Note that the area restored would likely be smaller than the areas drained in (a) or desired in (b).*

land), the concentration of organic matter, and the available surface area of plants and other substrates for the growth of microbes (Phipps and Crumpton 1994; Woltemade 2000). Residence time is a function of wetland size and the ratio of wetland area to watershed area. Some wetlands that receive agricultural runoff are effective even though this ratio is only 1:100 (Woltemade 2000).

Nutrient-removal functions have been simulated for watersheds (Crumpton and Baker 1993; Reiche 1994). In a 40 km^2 watershed in northern Germany, Trepel and Palmeri (2002) used a system based on GIS (geographic information system) to rank the most suitable areas for restoration, and explored alternative ways to predict N removal in the three wetlands that were judged the most suitable. In the most complex model, which used data on the watershed area, nitrate concentration, and residence time of water, their estimated N-removal efficiency was approximately 77%. Strategies for improving the quality of runoff from Upper Midwestern farms can build on such models. However, the wetlands that remove the most nutrients would not necessarily support the most species (Figure 2b). A wetland that collects heavy sediment loads and has a high influx of nutrients will probably develop monocultures of cattails or similar aggressive plants, rather than retaining a diversity of native plant species (Mensing *et al.* 1998; Galatowitsch *et al.* 1999; Werner and Zedler 2002; Woo and Zedler 2002). Nitrogen farms would also need a steady and moderate inflow of water, as opposed to the pulsed flood regime experienced by wetland reservoirs.

■ Enhancing biodiversity

Configuring wetland restoration projects for the benefit of plants and animals is more difficult than designing wetlands for flood abatement and improvement of water quality, because maintaining and increasing biodiversity is a complex task. While larger areas typically support more species, other factors, such as high habitat diversity, high productivity, or multi-layered plant canopies, can give the same result (Meffe and Carroll 1997). Connected habitats are essential for the dispersal and recolonization of restoration sites, and other factors become important for migratory species that require different habitats during nesting, spawning, or other seasons.

In native ecosystems, natural disturbances such as fires, floods, and water level fluctuations help sustain biodiversity by reducing competition between species and allowing smaller and shorter-lived species to survive. Appropriate disturbance regimes may therefore need to be included in the long-term management of restoration sites (Foster *et al.* 1998). However, disturbances that are either excessive, insufficient, or ill-timed can favor invasive species. We need to be able to distinguish between disturbance regimes that favor diversity and those that favor aggressive invaders. Sites with minimal human impacts on their hydrological and nutrient conditions can usually support more species (Mensing *et al.* 1998).

Complex feedback mechanisms should be considered when designing wetlands to maximize biodiversity. For example, diverse vegetation can trap more N (Callaway *et al.* unpublished), but this may reduce plant diversity (Bedford *et al.* 1999). While cattails might remove the most nutrients (Kadlec and Knight 1996), they can replace native species in the process (Woo and Zedler 2002). Ecologists have yet to determine how much N and

Figure 3. *McKenna Pond, a deepwater marsh near Madison, WI. River bulrush* (Scirpus fluviatilus) *dominates the emergent vegetation in many years, although reed canary grass is expanding at the pond edge, perhaps in response to inflows from adjacent agricultural land.*

P can be removed by wetlands without compromising plant and animal diversity (Figure 3).

The principles of conservation biology (eg Meffe and Carroll 1997) are relevant to the design process. The greatest biodiversity should result from restoring large wetlands that have mixed terrain and dispersal corridors, and are close to natural wetlands, ensuring that the appropriate disturbance regimes will occur indefinitely. However, we still do not know which of these factors is the most important in agricultural landscapes, and whether one feature might compensate for the lack of another. A large wetland with low plant diversity, for example, may support as many animal species as a smaller site with more plant species. There are few examples of the restoration of species-rich wetlands, so we should protect and enhance remnant wetlands in the watershed and increase the area of wetland to sustain broadly tolerant species. It therefore seems wise to choose restoration sites next to remnants of original habitats (Figures 2b, c; Olson and Harris 1997; Cedfeldt *et al.* 2000).

A few attempts have been made to rank areas with high potential for biodiversity restoration. Russell *et al.* (1997) used a GIS model to predict the restoration potential of sites along southern California's San Luis River. The authors calculated wetness from elevation and land cover from satellite imagery, and prioritized sites based on their size and whether they were connected to existing riparian habitat or open water. For the same river, Olson and Harris (1997) classified the suitability of river reaches for restoration based on land cover, land use, and connectivity data. Field sampling indicated the extent to which desired species needed to be restored and exotic plants removed. While these studies indicated which areas could be best restored by specific actions, the authors admit that the ideal combination of communities remains unknown. To determine this "desirable mix", "multiple resources and functions must be considered and overall objectives must be established" (Olson and Harris 1997).

Multiple wetland types for multiple functions

It is not likely that one restoration approach will maximize all ecosystem functions. Plans to restore multiple ecosystem services within watersheds must ideally consider a collection of sites (Figure 2c). We have to learn to predict functions of restored ecosystems based on simple metrics such as wetland position, size, and type, and then try to expand our predictions to multiple sites within watersheds and basins.

Several researchers have estimated the ecosystem services that might be regained by restoring multiple wetlands within large watershed basins. Hey *et al.* (2002) calculated the floodwater retention potential of six sites in the Upper Mississippi River Basin and concluded that restoring wetlands for use as temporary reservoirs could substantially reduce catastrophic flooding. For the 1 700 000 km^2 Baltic Sea basin, Jansson *et al.* (1998) estimated how much N existing wetlands could remove, and calculated that restoring all the drained wetlands would increase N-removal rates from 5–13% to 18–24%. In Maryland, Costanza *et al.* (2002) used their state-of-the-art model of the 2352 km^2 Patuxent River watershed to simulate the effect of returning all farmland to forest on plant productivity, water flow, and nutrient discharge. The most promising approach is that of Cedfeldt *et al.* (2000), who used GIS data to identify the 285 most functional wetlands within 21 326 km^2 of the Lake Champlain Basin, Vermont. The authors based the selection on multiple functions of biodiversity, water quality, and flood abatement. Although the approach was developed to identify existing wetlands whose preservation should be a high priority, it could be modified to identify opportunities to restore former wetlands.

While some planning for habitat restoration is underway along the Upper Mississippi River, we still lack clear criteria for designing watershed-scale restoration programs (Woltemade 2000). Learning how to configure wetland restoration projects to provide a desired mix of ecosystem services (Figure 3) would significantly advance our understanding of watershed management.

The ability to restore wetlands

Restored wetlands often fall short of their natural levels of biodiversity, functioning, and sustainability (Zedler 2000, NRC 2001). Typical shortcomings include water levels that are too high or low, plantings that die, animals that fail to use sites designed for them, and the presence of invasive exotic species, including some that damage vegetation (Figure 4). Some losses of function appear to be irreversible (Zedler and Callaway 1999). Although many former wetlands can be restored to a certain extent by filling drainage ditches and removing drainage tiles and levees, the outcome is often species-poor vegetation dominated by gener-

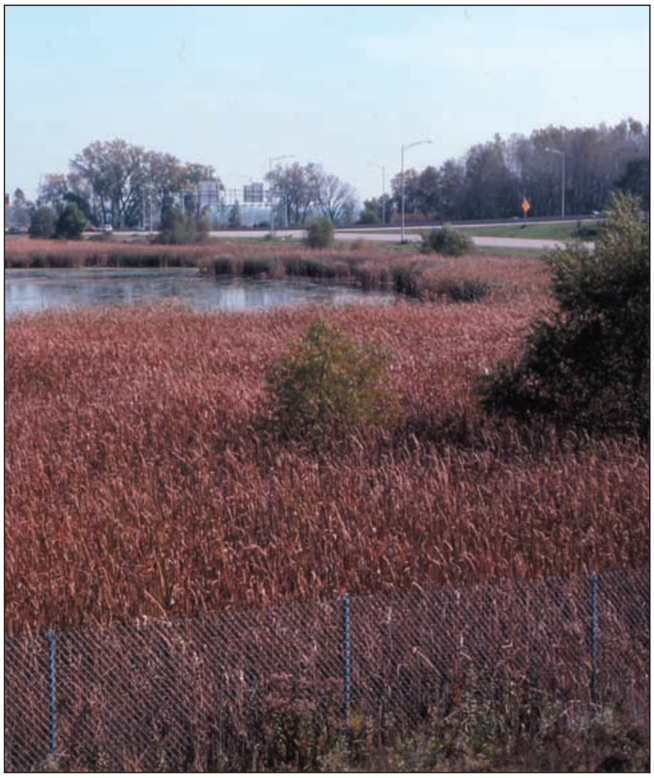


Figure 4. *Restored wetland in Madison, WI. Although designed for and planted with native sedge meadow plants, the site converted to cattails (*Typha *spp.) within 5 years.*

alists. Many efforts produce duck ponds ringed with cattails; fewer result in wetlands with the low-nutrient, seasonally rising, shallow waters required by many fens, sedge meadows, and wet prairies (Hunt *et al.* 1999; NRC 2001). The full restoration of biodiversity will require better and more diverse approaches and, in agricultural regions, landowners will have to understand that wetland services go far beyond waterfowl production and the aesthetics of open-water views (J Ruwaldt, pers comm).

The problems encountered at individual restoration sites are compounded when restoring functions at the watershed scale. It is particularly difficult to duplicate historical conditions in agricultural regions, where decades of cultivation have caused the land to subside, the soil quality to change, and the local pool of native species to shrink. Simply bringing the water back is not enough to support all the native species (Galatowitsch and van der Valk 1996; Hunt *et al.* 1999). Furthermore, agriculture will continue to dominate the Upper Midwestern landscape, and even the most intensive restoration efforts will not reestablish natural conditions. We therefore need to adjust our restoration goals at the outset and optimize the level of ecosystem services that can be accomplished in such "working landscapes". We should aim for structural and functional equivalence with reference ecosystems and watersheds, but will probably need to accept something less.

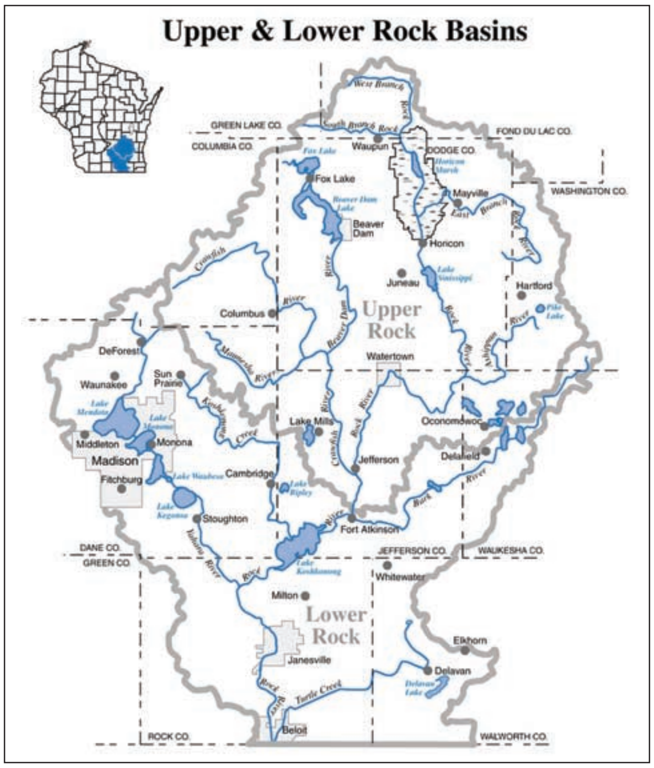
The task requires more than trial-and-error approaches (Zedler 2000). With hundreds of species to be sustained, thousands of tons of N to be trapped, and billions of dollars worth of flood damage to be prevented, we need to perform experiments that clarify the benefits of alternative restoration configurations and methods. We can learn while restoring if we adopt an adaptive management approach (Thom 1997; Walters 1997).

■ An adaptive approach

Adaptive management involves the application of alternative management actions, the appraisal of their effects, and the integration of these findings into future actions (Christensen *et al.* 1996). Applying this strategy to watershed restoration, managers and researchers ideally work together to (1) develop conceptual or GIS models relating alternative restoration configurations (such as those in Figure 2b) to the ecosystem services they provide, (2) provide annual assessments of biodiversity support, water quality improvement, and flood abatement at the watershed scale, (3) determine restoration effectiveness based on monitoring data, and (4) use these findings to improve models and revise restoration priorities. More effective watershed configurations should gradually emerge (Figure 2c). Because we do not know what the outcomes of these various restoration strategies will be, an adaptive approach becomes essential. It makes little sense to spend millions of restoration dollars every year without learning how to optimize the benefits.

Controlled experimentation would be ideal, but at this scale it is impractical at best, because watersheds are not good experimental replicates (each is unique), and because farmers are unlikely to offer up their lands for restoration at the request of scientists. Yet it may be possible to select a few similar watersheds within a basin and identify key sites for setting up alternative restoration configurations, with the aim of finding two or three watersheds that differ primarily in the types and extent of restoration. Agencies could offer more money for desired parcels of land, and inform key landowners of the high costs and low benefits that accompany the cultivation of wetland – information that is easily calculated from existing precision-agriculture models (J Norman, pers comm). The resulting "demonstration watersheds" could then be compared in terms of ecosystem services. Other watersheds could be added to the study, to compare the "first-come first-served" and "bigger-is-better" approaches to implementing Farm Bill restoration programs.

A revolutionary watershed-scale experiment is needed to find the best ways to accomplish these goals, and there is nothing to stop us from establishing demonstration watersheds with alternative restoration configurations immediately. Wisconsin's Rock River Basin (Figure 5), with over two dozen watersheds, is an ideal starting place. Many of the watersheds within the basin could be targeted for wetland restoration using monies available to the state under the Farm Bill, with the intention of reducing nutrient discharges, flooding, and biodiversity loss. When the results of restoring wet-



Courtesy of the Rock River Coalition

Figure 5. *The Wisconsin portion of the Rock River Basin, noted for its high concentrations of N and P and suitable for watershed-scale restoration projects. Demonstration watersheds could be selected for establishing alternative restoration configurations (see Figure 2b), as part of an adaptive management program leading to informed choices for restoration of multiple ecosystem services at the watershed scale. (Developed by the University of Wisconsin-Extension Environmental Resources Center).*

lands in different ways become clearer, we could be more strategic in the implementation of Farm Bill programs. The benefits of an adaptive approach to restoring wetland functions at the watershed scale might not be obvious for a decade or more, but the rewards are exciting to imagine.

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