Potential Benefits of Wetland Filters for Tile Drainage Systems: Impact on Nitrate Loads to Mississippi River Subbasins

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Nontechnical Summary

The primary objective of this project was to estimate the nitrate reduction that could be achieved using restored wetlands as nitrogen sinks in tile-drained regions of the upper Mississippi River (UMR) and Ohio River basins. This report provides an assessment of nitrate concentrations and loads across the UMR and Ohio River basins and the mass reduction of nitrate loading that could be achieved using wetlands to intercept nonpoint source nitrate loads. Nitrate concentration and stream discharge data were used to calculate stream nitrate loading and annual flow-weighted average (FWA) nitrate concentrations and to develop a model of FWA nitrate concentration based on land use. Land use accounts for 90% of the variation among stations in long term FWA nitrate concentrations and was used to estimate FWA nitrate concentrations for a 100 ha grid across the UMR and Ohio River basins. Annual water yield for grid cells was estimated by interpolating over selected USGS monitoring station water yields across the UMR and Ohio River basins. For 1990 to 1999, mass nitrate export from each grid area was estimated as the product of the FWA nitrate concentration, water yield and grid area. To estimate potential nitrate removal by wetlands across the same grid area, mass balance simulations were used to estimate percent nitrate reduction for hypothetical wetland sites distributed across the UMR and Ohio River basins. Nitrate reduction was estimated using a temperature dependent, area-based, first-order model. Model inputs included local temperature from the National Climatic Data Center and water yield estimated from USGS stream flow data. Results were used to develop a non-linear model for percent nitrate removal as a function of hydraulic loading rate (HLR) and temperature. Mass nitrate removal for potential wetland restorations distributed across the UMR and Ohio River basin was estimated based on the expected mass load and the predicted percent removal. Similar functions explained most of the variability in per cent and mass removal reported for field scale experimental wetlands in the UMR and Ohio River basins. Results suggest that a 30% reduction in nitrate load from the UMR and Ohio River basins could be achieved using 210,000-450,000 ha of wetlands targeted on the highest nitrate contributing areas.

POTENTIAL BENEFITS OF WETLAND FILTERS FOR TILE DRAINAGE SYSTEMS: IMPACT ON NITRATE LOADS TO MISSISSIPPI RIVER SUBBASINS

Introduction

Agricultural applications of fertilizers and pesticides have increased dramatically since the middle 1960s and the impact of agrochemicals on water quality has become a serious environmental concern. Nitrate is a particular concern because of (1) the widespread use of nitrogen in modern agriculture, (2) the high mobility of nitrate in surface and groundwater, and (3) the potential adverse impacts on both public health and ecosystem function. Annual application of fertilizer-N in the U.S. has grown from a negligible amount prior to World War II to approximately ten million metric tons of N per year (Terry and Kirby, 1997). Cultivated croplands can lose a significant amount of nitrogen in agricultural drainage water, primarily in the form of nitrate (Neely and Baker, 1989). The impacts of chemical intensive agriculture are a special concern in the U.S. Corn Belt. Non-point source nitrogen loads to surface waters in the region are among the highest in the Mississippi River Basin. In addition to the potential local impacts on receiving waters in the Corn Belt, nitrogen loads from the region are suspected as a primary source of nitrate contributing to hypoxia in the Gulf of Mexico.

The problem of excess nitrate loads can probably be ameliorated by a combination of in field and off site practices, but the limitations and appropriateness of alternative practices must be considered. Nitrate is transported from crop land primarily in subsurface drainage, especially in extensively tile drained areas like the Corn Belt (Figures 1 and 2). As a result, grass buffer strips, woody riparian buffers, and many other practices suited to surface runoff have little opportunity to intercept nitrate loads in these areas. Studies suggest that better nutrient management has some potential to reduce nitrate losses from crop land, but that potential is probably limited to 25% or less (Baker et al. 1997). Wetlands sited to intercept tile drainage have the potential to significantly reduce nitrate loads, and this approach is particularly promising for heavily tile drained areas like the Corn Belt. This region was historically rich in wetlands, and in many areas, farming was made possible only as a result of extensive drainage. As a result, there are opportunities for wetland restoration throughout the region and there is considerable potential for restored wetlands to intercept tile flow.

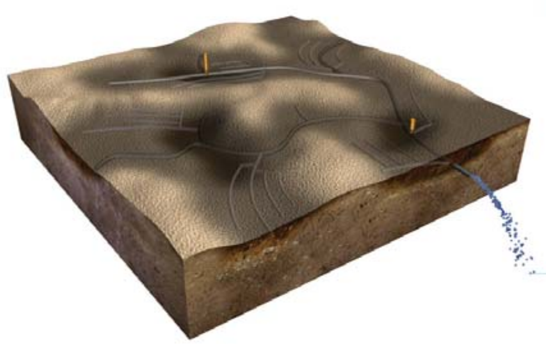

Figure 1. Tile drainage system for a typical agricultural landscape of the western Corn Belt

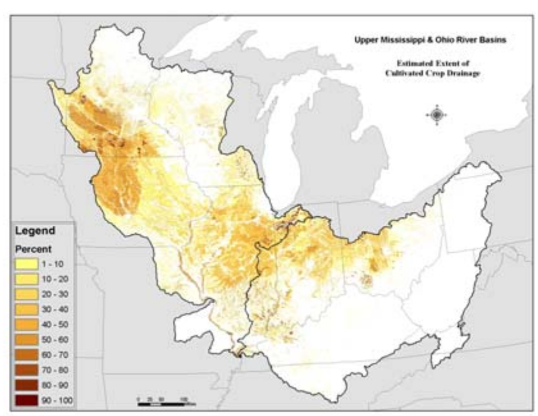

Figure 2. Estimated extent of agricultural drainage based on soils and land use.

The primary objective of this project was to estimate the nitrate reduction that could be achieved using restored wetlands as nitrogen sinks in tile-drained regions of the upper Mississippi River (UMR) and Ohio River basins. This report provides an assessment of nitrate concentrations and loading in the UMR and Ohio River basins, the effect of land use on surface water nitrate concentrations in this region, and the distribution of tile drained areas across the Corn Belt. Tile drained areas are the major source of nitrate loading and are candidates for restoring wetlands to intercept tile drainage and reduce nitrate loads to receiving waters. For suitable candidate areas, we estimate both the total mass reduction of nitrate and the percentage reduction of nitrate loading that could be achieved using wetlands to intercept tile drainage.

A Performance Based Approach to Wetland Restoration

Since the mid-1980s, a variety of state and federal programs have been used to promote wetland restoration. These continuing efforts provide a unique opportunity for control of agricultural non-point source pollution. However, wetland restorations have been motivated largely by concern over waterfowl habitat, and site selection criteria have not primarily addressed water quality functions. Of more than 500 wetland restorations in the southern prairie pothole region surveyed by Galatowitsch (1993), most drain very small areas and intercept insufficient contaminant loads to significantly affect water quality. The effect of wetlands on watershed scale nitrate reduction is obviously constrained by the fraction of total nitrate load that the wetlands intercept. If not sited so as to intercept a significant fraction of the watershed load, restored wetlands can have little effect on either nitrate concentration or mass nitrate export at the watershed scale. This does not lessen the promise of wetlands for water quality improvement but rather underscores the need for targeted, performance based approaches to wetland restorations (Crumpton 2001). The Iowa State University Wetlands Research Group has worked to extend the application of performance forecast models to siting, design, and assessment of wetland restorations in agricultural watersheds (Crumpton and Baker 1993; Crumpton et al. 1995; Baker et al. 1997; Crumpton 2001, 2005). Results from experimental wetlands were used to develop a general model of nitrate loss for wetlands receiving non-point source nitrate loads and this model was validated against field data for research sites in Illinois and Iowa. The nitrate loss model was then combined with hydraulic and nitrate loading estimates to simulate nitrate loading and loss for wetlands in agricultural watersheds and evaluate the water quality benefits of wetland restoration (Crumpton et al. 1995; Crumpton 2001). This work provided much of the research foundation for the Iowa Conservation Reserve Enhancement Program, a targeted, performance-based strategy for reducing nitrate loads from tile-drained landscapes.

Created by a partnership between the Iowa Department of Agriculture and Land Stewardship and the U.S. Department of Agriculture, the Iowa CREP provides financial incentives to landowners to restore wetlands that intercept tile drainage from agricultural watersheds. The Iowa CREP was approved on August 17, 2001, and is available in thirty-seven counties covering the most intensively tile drained region of North-Central Iowa (Figure 3). To be eligible for Iowa CREP funding, (1) wetlands must be restored below a tile-drainage system that drains at least 200 ha (500 acres) of primarily cropland, (2) wetland area must be between 0.5 and 2% of the upslope contributing drainage area, (3) at least 75% of the wetland pool area must be less than 0.9 m deep so as to encourage establishment of emergent vegetation, and (4) wetlands must be designed so as to assure the vested drainage rights of upstream landowners. A total of 20 Iowa CREP wetlands have been constructed to date (Figure 3), ranging in size from 1.4 to 7.5 ha. These 20 wetlands intercept flows from drainage areas ranging from 208 to 1478 ha and span the 0.5% - 2% range in wetland/watershed area ratio set by the program criteria (program data provided by Iowa Department of Agriculture and Land Stewardship).

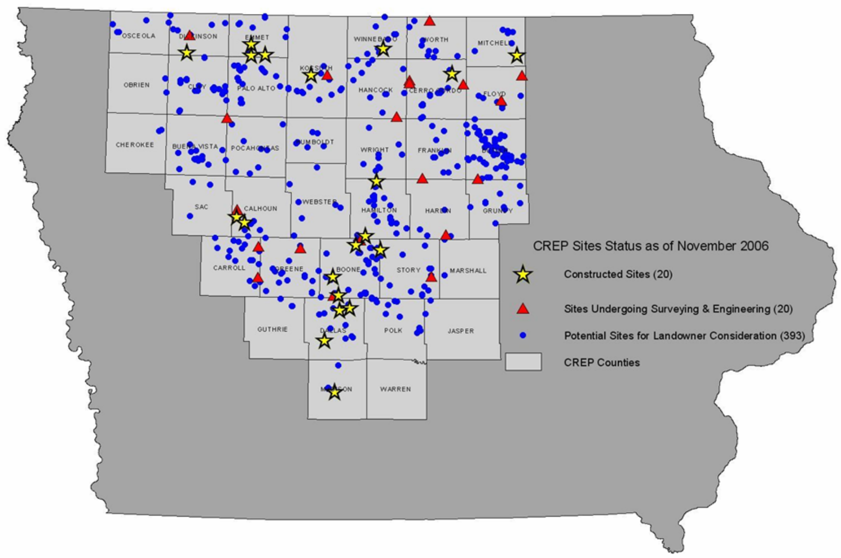

Figure 3. Counties eligible for IA CREP funding and status of Iowa CREP sites. (Figure provided by Iowa Department of Agriculture and Land Stewardship).

A unique aspect of the Iowa CREP is that nitrate reduction is not simply assumed based on wetland acres enrolled, but calculated based on the measured performance of CREP wetlands. As an integral part of the Iowa CREP, representative wetlands are monitored each year to document nitrate reduction. The wetlands selected for monitoring include CREP wetlands as well as wetlands restored under other programs but still meeting the CREP program criteria. This allows monitoring of some wetlands that have been in place much longer than CREP program wetlands. By design, the wetlands selected for monitoring span the 0.5% - 2% wetland/watershed area ratio range approved for Iowa CREP wetlands. The wetlands also span a range in average nitrate concentration from less than 10 mg/l (Hanlontown Slough) to approximately 30 mg/l (Finley Wetland). The wetlands thus provide a broad spectrum of those factors most affecting wetland performance: hydraulic loading rate, residence time, nitrate concentration, and nitrate loading rate. In addition to weekly grab samples, a subset of wetlands is instrumented with automated samplers and flow meters at wetland inflows and outflows. Water levels are monitored continuously at outflow structures in order to calculate changes in pool volume and discharge. Starting in 2006, wetland water temperatures are recorded continuously for modeling nitrate loss rates. (Prior to 2006, water temperatures were estimated based on air temperature.)

Despite significant variation with respect to average nitrate concentrations and loading rates, the wetlands display similar seasonal patterns. Nitrate concentrations and mass loads are typically highest during high flow periods in spring and early summer, and decline with declining flow in late summer and fall. Figure 4 illustrates the seasonal patterns in nitrate concentrations and flows for four wetlands spanning a range of hydraulic loading rates (HLRs to Hendrickson Marsh < van Horn Wetland < Louscher Wetland < Triple I Wetland) and wetland:watershed area ratios (Table 1). Each of these wetlands has a single major inflow and discharges at a single outflow with a control structure. The inflow to each wetland is the combined surface and subsurface discharge from a drainage district of at least 200 ha in size planted primarily to corn and soybean. Hendrickson Marsh, van Horn Wetland, and Louscher Wetland follow the typical pattern for Iowa CREP wetlands with higher flows, concentrations, and nitrate loads in spring and early summer (Figure 4). Flows, nitrate loads and to a lesser extent nitrate concentrations decline after late summer and remain low through the remainder of the season. Inflows to Hendrickson Marsh, van Horn Wetland, and Louscher Wetland also display similar patterns with respect to variability in nitrate concentrations in response to flow variability. Nitrate concentrations in the inflows to these wetlands tend to be quite stable except for brief declines in concentration coinciding with some but not all flow events. The brief declines are probably a result of dilution by surface runoff water. In contrast, nitrate concentrations at the inflow to Triple I Wetland are much more variable and tend to rise in response to most flow events. The difference may well be related to differences in soils, topography, geomorphology, and/or drainage systems, but this has not yet been examined further.

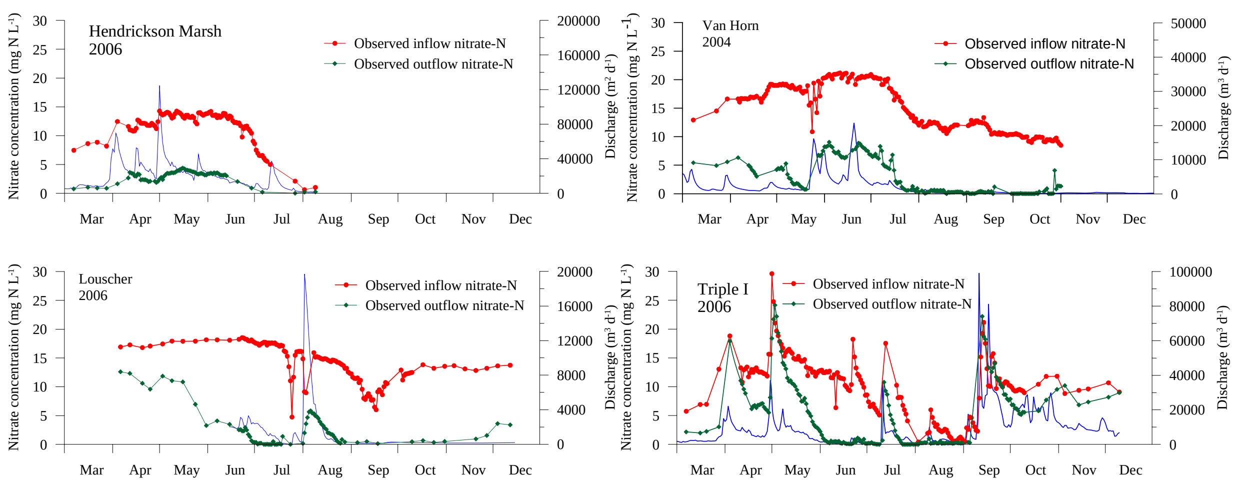

Figure 4. Nitrate concentrations and flows for "CREP" wetlands with different hydraulic loading rates (Hendrickson Marsh < van Horn Wetland < Louscher Wetland < Triple I wetland).

For Hendrickson Marsh, van Horn Wetland, and Triple I Wetland, mass nitrate loads and mass nitrate export were calculated based on the daily flow and concentration data for wetland inflows and outflows and summed to calculate annual mass balances (Table 1, based on estimated flows Louscher Wetland would have had mass loss rates similar to those measured at Triple I and higher % loss). These three wetlands were selected for calculating annual mass balances because monitoring was initiated soon enough after thaw to capture spring flows and continued through the season unless flows ceased. Because of delays in deploying monitoring equipment, annual mass balances for Louscher could not be calculated. In 2006, Triple I Wetland experienced rare, late season flooding that delivered the equivalent of a normal year's flow within a few weeks. Triple I mass balances were calculated both for the entire season and for the period prior to the late season flood. Although the results for the late season flood fit the same functions as the remaining data (Figure 6), the hydraulic and nitrate loading rates are double those of any of the other systems considered and the mass loss rates measured are probably much higher than could reasonably be expected for most systems. The flow to Triple I prior to the late season flood was near the 10 year average annual flow expected for this wetland. Hendrickson Marsh was drained for vegetation management after flows had declined to seasonal lows in August. As in the case of Louscher and van Horn, flows from the Hendrickson Marsh watershed remained low for the rest of the field season and would have contributed little to the annual mass balance had the wetland not been drained. Nitrate losses in seepage estimated based on volumetric seepage coefficients and nitrate concentrations were not a significant component of the nitrate budget (less than 7% at Hendrickson and less than 4% at van Horn and Triple I). These are lower rates of seepage loss than reported for many of the wetlands in the analyses that follow (Table 2 and Figures 7-10), but unlike most of those wetlands, the IA CREP wetlands are not built alongside rivers but rather at or above the headwaters of small streams. The stream begins as the wetland outflow. In this respect, the CREP wetlands are more like Eagle Lake Marsh (Davis et al. 1981) or the in stream wetland described by Hunt et al. (1999). Annual mass balance results for Hendrickson Marsh, van Horn Wetland, and Triple I Wetland are summarized in Table 1. (*Figure 6 includes the annual mass balance results from Table 1 for Hendrickson Marsh, van Horn Wetland, and Triple I Wetland as well as mass balance results for the period prior to the late season flood at Triple I Wetland. Results of the late season flood at Triple I Wetland are not included in the subsequent analyses represented in Figures 7-10.)*

In support of the CREP monitoring program, mass balance modeling was used to estimate the variability in performance of CREP wetlands that would be expected due to spatial and temporal variability in temperature and precipitation patterns. The percent nitrate removal expected for CREP wetlands was estimated based on hindcast modeling over the 10 year period from 1996 through 2005. Nitrate removal was modeled as a temperature-dependent first-order process (Crumpton 2001). The range of outflow concentrations predicted for Triple I Wetland (a high loading rate site) and Hendrickson Marsh (a low loading rate site) based on modeling with 2006 inputs and forcing functions are illustrated in Figure 5 along with the observed concentrations and flows. The seasonal patterns of measured and modeled outflow concentrations show reasonable correspondence over the very broad range of flow conditions represented by these two sites. Comparison of the 10 year hindcast modeling results with the percent nitrate removal measured for three Iowa wetlands (Table 1) also illustrates reasonably good correspondence between observed and modeled performance of the wetlands (Figure 6).

Table 1. Nitrate mass balance, concentration and hydraulic load data for selected Iowa wetlands.

Wetland & Year	Wetland to watershed area ratio %	Load (kg N ha^{-1})	Removal (kg N ha^{-1})	Percent Removal	FWA Conc. (mg N L^{-1})	HLR (m)
van Horn, 2004	2.25	1314	897	68	18.0	7.3
Hendrickson Marsh, 2006	2.16	469	368	78	11.8	4.0
Triple I, 2006 pre-flood	0.57	3807	1510	40	13.0	29.6
Triple I, 2006 including late season flood	0.57	9240	2310	25	11.9	78

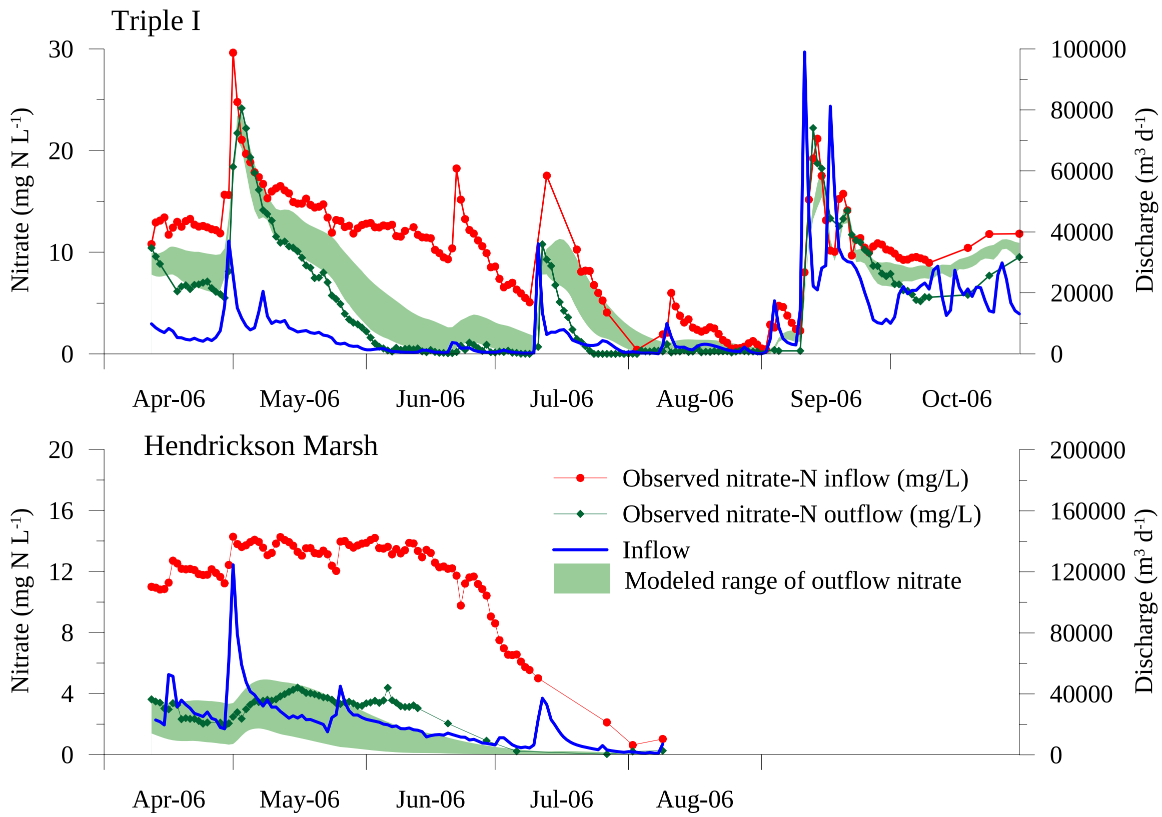

Figure 5. Measured and modeled nitrate concentrations and flows for Triple I Wetland and Hendrickson Marsh in 2006.

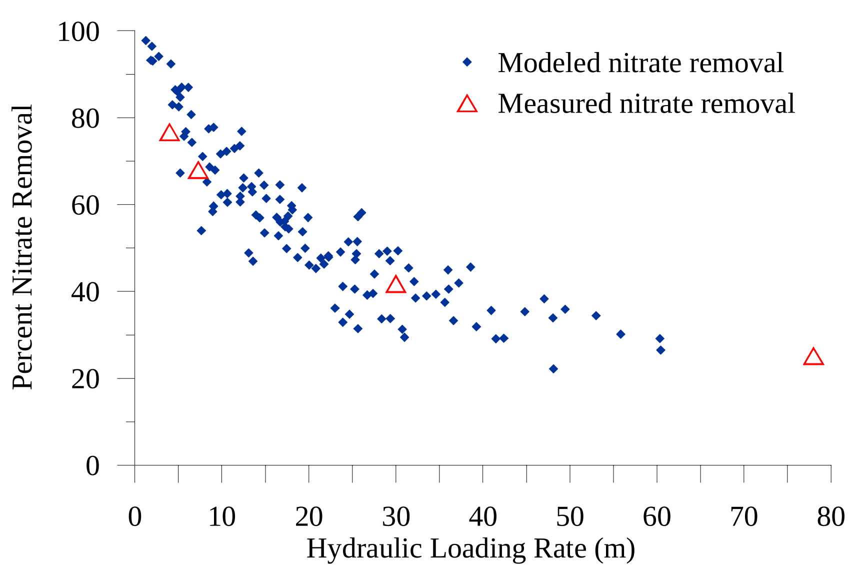

Figure 6. Modeled nitrate removal efficiencies for CREP wetlands based on 1996 to 2005 input conditions and measured nitrate removal efficiencies for CREP wetlands in 2004 & 2006.

Based on both the hindcast modeling results and on the measured performance of CREP wetlands, percent nitrate removal by CREP wetlands is clearly a function of hydraulic loading rate (Figure 6). The importance of hydraulic loading rate is confirmed by analysis of nitrate removal rates reported for wetlands in the UMR and Ohio River basins. Based on 34 "wetland years" of available data (12 wetlands, 1-9 years of data each; Table 2) for sites in Ohio (Mitsch et al 2005; Zhang and Mitsch 2000, 2001, 2002, and 2004), Illinois (Hey et al. 1994; Kovacic et al 2000; Phipps and Crumpton 1994; Phipps 1997), and Iowa (Table 1, this report; Davis et al 1981), percent mass nitrate removal is clearly related to hydraulic loading rate (Figure 7). When the analysis is restricted to only those wetlands meeting the 1 ha minimum size requirement for the IA CREP, a similar relationship is found but with slightly higher percent removal rates.

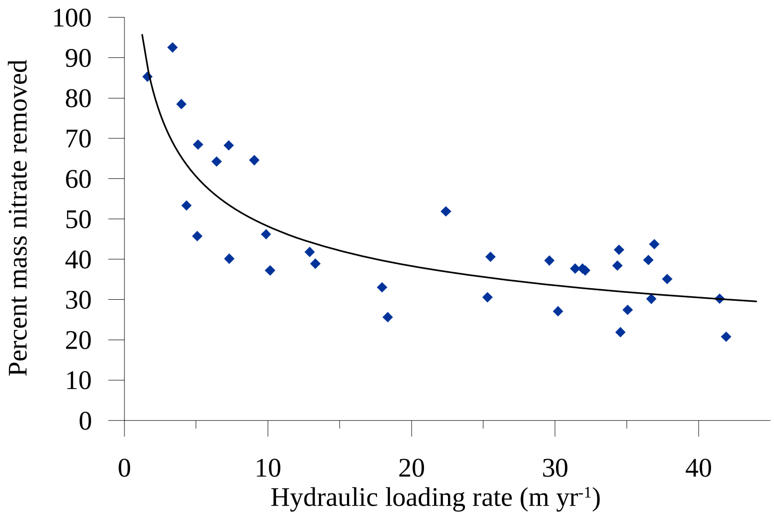

Figure 7. Percent mass nitrate removal in wetlands as a function of hydraulic loading rate. Best fit for percent mass loss = 103 × (annual hydraulic loading rate)$^{-0.33}$ (R^2 = 0.69).

In contrast to percent removal, hydraulic loading rate explains relatively little of the pattern in nitrate mass removal rates. Although mass removal will obviously be constrained at lower HLRs (because the mass load decreases with decreasing HLR), mass removal rates vary widely at higher HLRs. Mass nitrate removal rates can vary considerably more than percent nitrate removal among wetlands receiving similar hydraulic loading rates. Mass removal rates are the product of percent removal, hydraulic loading rate (HLR), and flow-weighted average (FWA) concentration, and as such include the variability in each of these. However, much of the variability in mass nitrate removal can be accounted for by explicitly and separately considering the effect of HLR and FWA concentration. For the wetlands considered here, mass nitrate removal rate = [103 × (HLR)$^{-0.33}$] × HLR × [FWA nitrate concentration] × [unit conversion factors]. Combining terms and incorporating unit conversion factors yields the function:

Mass nitrate-N removed = 10.3 × (HLR)$^{0.67}$ × FWA nitrate-N concentration

Where mass nitrate removal is in kg N ha^{-1} yr^{-1}

HLR is in m yr^{-1} and

FWA nitrate concentration is in g N m^{-3}.

A comparison of the measured and predicted nitrate removal for these wetlands demonstrates that the performance of wetlands representing a broad range of loading and loss rates can be reconciled by a model explicitly incorporating hydraulic loading rates and nitrate concentrations (Figure 8). This relationship can be further illustrated (Figure 6) by fitting the observed wetlands data to a surface plot of the mass nitrate removal function. The isopleths on the function surface illustrate the combinations of HLR and FWA that can be expected to achieve a particular mass loss rate. The function described above explains 94 % of the variability in mass removal rates for the wetlands considered here (Table 2, Figures 8 & 9).

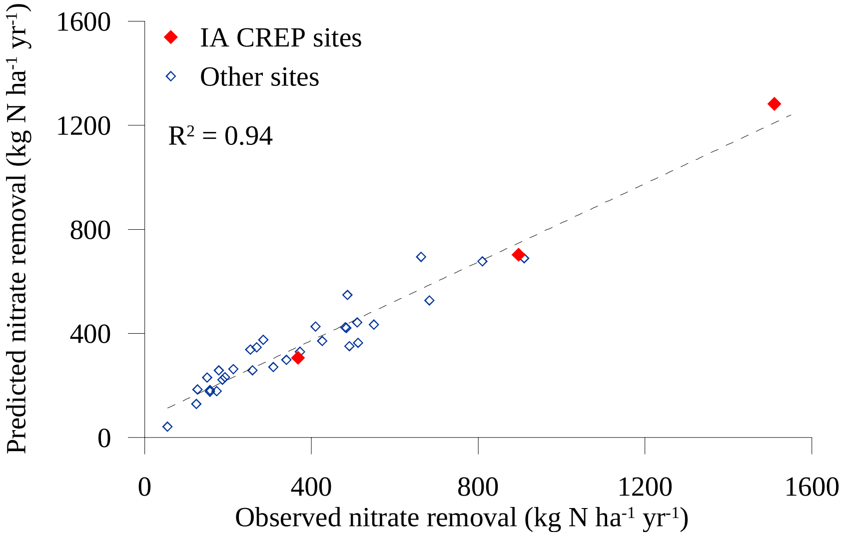

Figure 8. Observed nitrate mass removal in wetlands versus removal predicted from HLR and FWA nitrate concentrations. Predicted mass nitrate removed = 10.3 × $(HLR)^{0.67}$ × FWA.

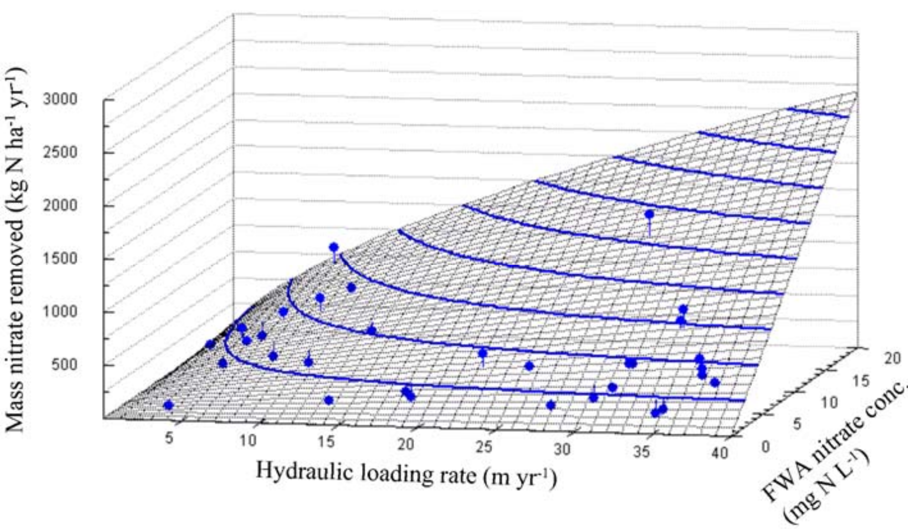

Figure 9. Observed nitrate mass removal in wetlands (points) versus removal rates predicted from HLR and FWA nitrate concentrations (surface). Predicted mass nitrate removed = 10.3 × $(HLR)^{0.67}$ × FWA.

Table 2. Nitrate mass inflow, outflow, HLR, FWA nitrate concentration, and % mass removal for wetlands in the Upper Mississippi and Ohio River basins (data source as indicated).

SOURCE	Inflow nitrate kg N/ha/yr	Outflow nitrate kg N/ha/yr	HLR m/yr	FWA nitrate mg N/l	Percent removed
Mitsch et al. 2005; Zhang and Mitsch 2000, 2001, 2002, 2004 (Olentangy River Wetland Research Park, Ohio)					
Wetland 1 1994	572	416	35.0	1.63	27.3
Wetland 2 1994	579	452	34.6	1.68	21.9
Wetland 1 1995	858	679	41.9	2.05	20.9
Wetland 2 1995	858	599	41.5	2.07	30.2
Wetland 1 1996	584	391	18.0	3.25	33.0
Wetland 2 1996	585	435	18.3	3.19	25.6
Wetland 1 1997	2110	1300	34.4	6.14	38.4
Wetland 2 1997	2150	1240	34.5	6.24	42.3
Wetland 1 1998	1360	950	36.7	3.71	30.1
Wetland 2 1998	1380	830	36.5	3.78	39.9
Wetland 1 1999	786	573	30.2	2.60	27.1
Wetland 2 1999	819	510	31.4	2.61	37.7
Wetland 1 2000	1293	812	32.1	4.03	37.2
Wetland 2 2000	1284	800	31.9	4.03	37.7
Wetland 1 2001	1122	631	36.9	3.04	43.8
Wetland 2 2001	1062	689	37.8	2.81	35.1
Wetland 1 2003	1049	623	25.5	4.11	40.6
Wetland 2 2003	987	475	22.4	4.41	51.9
Kovacic et al. 2000 (Champaign County, Illinois)					
Wetland A 1995	623	338	5.1	12.26	45.7
Wetland A 1996	1435	772	9.9	14.54	46.2
Wetland A 1997	1058	375	9.1	11.69	64.6
Wetland B 1995	350	163	4.3	8.08	53.3
Wetland B 1996	793	283	6.4	12.33	64.3
Wetland B 1997	497	157	5.1	9.68	68.5
Wetland D 1995	633	379	7.3	8.66	40.1
Wetland D 1996	1249	763	13.3	9.39	38.9
Wetland D 1997	723	454	10.2	7.12	37.2
Phipps and Crumpton 1994; Phipps 1997 (Des Plaines River Wetlands Demonstration project, Illinois)					
Wetland 3 1992	468	325	25.3	1.85	30.6
Wetland 4 1992	59.1	4.4	3.36	1.76	92.6
Wetland 5 1992	282	164	12.9	2.19	41.8
Davis et al. 1981					
Eagle Lake Marsh, Iowa 1979	203	30	1.6	12.52	85.3
This report (Iowa CREP monitoring, Des Moines Lobe, IA)					
van Horn Wetland 2004	1314	417	7.3	18.05	68.2
Hendrickson Marsh 2006	469	101	4.0	11.78	78.5
Triple I Wetland 2006 (pre flood)	3807	2297	29.6	12.86	39.7

The relationship between HLR, FWA, and mass loss was used to guide selection of CREP wetlands for monitoring in 2007. Wetlands were chosen in part to more uniformly populate the function relating mass loss rate to HLR and FWA (Figure 10) and provide a stronger basis for site selection and design of wetland restorations.

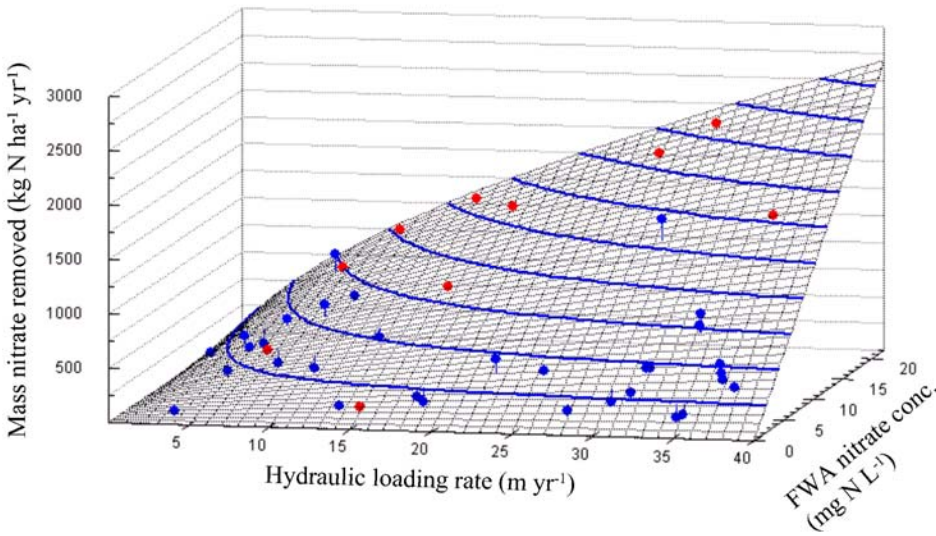

Figure 10. Nitrate mass removal rates predicted from HLR and FWA nitrate concentrations (surface) versus measured nitrate mass removal (blue points). Red points represent the predicted nitrate mass removal for CREP wetlands selected for monitoring in 2007 (based on their estimated HLR and FWA). Predicted mass nitrate removed = 10.3 × $(HLR)^{0.67}$ × FWA.

Even with a reasonable model of wetland performance, predicting nitrate reduction requires an estimate of site specific hydrologic and nitrate loading rates. These can vary considerably for different locations across the Corn Belt as well as from year to year at any given location.

Nitrate Concentrations and Loads in the Upper Basin

Nitrate concentration and stream discharge data were obtained from the U.S. Geological Survey (USGS) National Stream Quality Accounting Network (NASQAN) and National Water Information System (NWIS) for selected gage/sampling stations in the UMR and Ohio River basins. Nitrate concentration measurements for these monitoring stations were made at approximately monthly intervals. In order to estimate annual nitrate loads, estimated nitrate concentrations are needed for days when no sample result is available. In this work, several techniques were used to accomplish this. One method involved using a linear interpolation between successive values to estimate a concentration for each day. Another technique involved using one of the statistical regression methods available in the USGS (2004) LOADEST software to estimate nitrate concentrations as a function of linear and quadratic terms for the logarithm of discharge, a temporal trend, and cyclic terms to account for seasonal variation. Daily concentrations values were multiplied by the daily measured discharge to obtain a daily nitrate mass load. Daily nitrate loads were summed to estimate monthly and annual nitrate loadings.

The nitrate loads determined from USGS data were used in several ways in this work. First, monthly loads were used to assess seasonal variation in nitrate loading for the UMR, Missouri River, and Ohio River. Second, annual nitrate loads determined by the linear interpolation method were divided by annual discharge at NASQAN stations to obtain flow-weighted average (FWA) nitrate-N concentrations. These FWA nitrate-N concentrations were used to develop a model for estimating FWA nitrate concentrations based on land use. This model was subsequently used to estimate nitrate concentrations in a GIS model for nitrate loading across the UMR and Ohio River basins. Last, annual nitrate loads determined by the regression method for selected USGS stations were compared with the GIS nitrate loading model to validate the performance of the GIS model.

Estimating Flow-Weighted Average Nitrate Concentrations for the Upper Mississippi and Ohio River Basins

NASQAN nitrate concentration data are available for stations across the UMR and Ohio River basins at an approximately monthly sampling interval from 1973 through 1994 (Alexander et al. 1998), although sampling frequency varied over time at some stations. In general, daily discharge data are available. Estimated annual nitrate loading values determined by a linear interpolation method were divided by annual discharge to obtain annual FWA nitrate concentrations for each measurement station.

Percent land use data for 1987 urban land, crop land, pasture land, forest land, range land, farm land, and other land available on the NASQAN web pages were used to determine a relationship between average flow-weighted nitrate concentration and land use for selected sites in the UMR and Ohio River drainage basins. Sites with large upstream reservoirs, extensive upstream urban areas, or with a watershed area less than 200 square miles were excluded. A non-linear model with percent cropland as the explanatory variable was found to provide a good fit to the data using a least sum of squared errors criterion. The model accounts for 90% of the observed variation in the average of 1980 to 1993 annual FWA nitrate concentrations from 52 stations (Figure 11). For comparison, reduced nitrogen (calculated as total nitrogen minus nitrate) shows a slight, but statistically significant, increase with percent crop land (Figure 11).

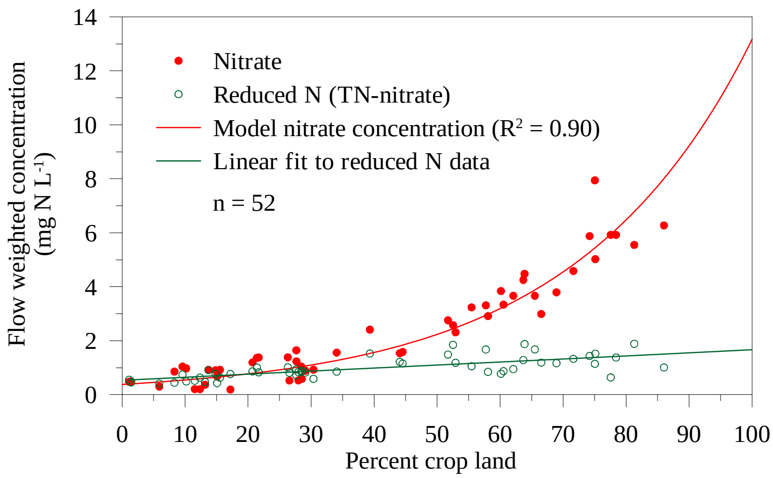

Figure 11. Flow-weighted average nitrate and reduced N versus percent cropland.

The regression model developed for NASQAN sites (Figure 11) was combined with land use data to estimate FWA stream nitrate concentrations across UMR and Ohio River basins. First, a grid of percent crop land was created based on spatial analysis of 1992 Landsat land cover data for the UMR and Ohio River basins (Figure 15c). Then, the regression model developed for NASQAN sites was applied to derive an estimate of nitrate concentration for each grid cell (Figure 15a).

For comparison, nitrate concentrations were also estimated based on U.S. EPA STORET data. Surface water nitrate and nitrite + nitrate data were downloaded from the U.S. EPA STORET data warehouse for states in the Missouri River, UMR, and Ohio River basins. Because there was considerably less data in the STORET system from Indiana, Missouri, and Ohio relative to other states of interest, additional data for these states was obtained directly from state agencies. Nitrate and nitrite + nitrate data were both treated as nitrate and concentrations were expressed as nitrate-N in mg/L. Surface water nitrate concentrations in the date range 1990 to 2005 were utilized in this work. Some samples were deemed to be unsuitable for reasons including unusually high concentration values (i.e. gross outliers), missing sampling date, missing or erroneous latitude and longitude data, or the sample was collected immediately downstream of water treatment or industrial discharge sites. Samples utilized were collected primarily from streams or rivers.

Because flow data suitable for calculating FWA concentrations are not available in the STORET database, the USGS NASQAN FWA nitrate concentrations were compared to monthly arithmetic average concentrations to determine a suitable measure and time frame for a surrogate of FWA concentration. Because we are primarily interested in developing a surrogate of FWA concentrations for tile-drained agricultural lands with generally elevated nitrate concentrations, for this analysis three NASQAN stations with FWA concentrations less than 0.3 mg/L nitrate-N were excluded leaving 49 stations in the Upper Mississippi and Ohio River basins for these comparisons. The ratio of the monthly average concentrations to the average 1980 to 1993 annual FWA were plotted using box-plots for each month (Figure 12). These box-plots indicate that the monthly averages for December through June are generally within about 20% of the FWA more than 50% of the time, while the bulk of the July to November averages tend to be less than the FWA. Nitrate loads in the Ohio River near Grand Chain, IL tend to increase from November to March, decrease from March to July and are generally low from July to November (Figure 13). UMR nitrate loads above the confluence of the Mississippi River and Missouri River tend to increase from November to May, decrease from June to September and remain low during the fall. The peak UMR load months are generally April to June, about two months after the peak February to April Ohio River load months. The Missouri River monthly average nitrate loads generally closely follow the UMR loads, but are generally lower than the UMR or Ohio River loads. Water flow generally peaks in February, March and April in the Ohio River Basin at Dam 53 near Grand Chain, IL and about two months later, in April, May and June, in the Missouri River at Hermann, MO and the UMR (Mississippi River at Thebes, IL minus Missouri River at Hermann, MO) (Figure 14). Coupling these periods of peak nitrate load and water flow with the time frame over which the monthly average concentrations are closest to the annual FWA and avoiding the winter months when the soil may be frozen in the northern regions of the UMR basin suggests that the average of the March to June nitrate concentrations should approximate the long term average annual FWA concentration reasonably well. Accordingly, the March to June time period was selected for further analysis of the STORET data.

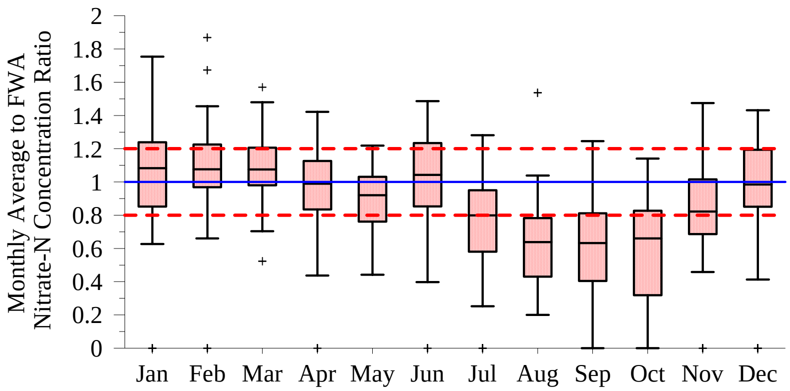

Figure 12. Ratio of average monthly nitrate concentration to average annual FWA nitrate concentration for 49 NASQAN stations from 1980 to 1993.

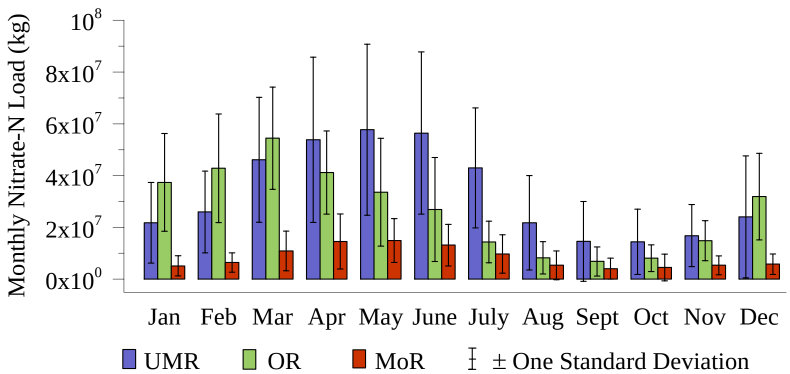

Figure 13. Monthly average nitrate load from 1973 to 2004.

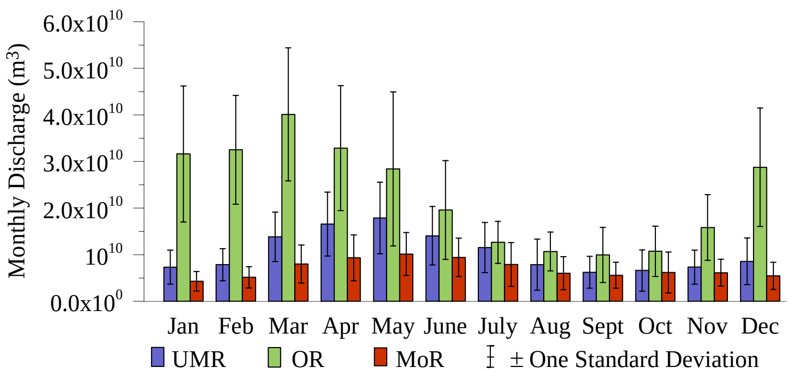

Figure 14. Monthly average discharge from 1973 to 2004.

The median value for samples collected during March to June from each STORET data station during 1990 to 2005 was determined (70,424 sample results). The median value was used because of occasional outlying data values sufficiently removed from the bulk of the data to adversely impact the arithmetic average. These “outliers” were almost always associated with stations having relatively low concentrations and the regional concentration plots were not significantly affected by use of the median rather than the average. The median values at each station latitude and longitude were contoured to illustrate the magnitude and spatial distribution of nitrate concentrations in surface waters across the UMR and Ohio River basins. Stations were included only if at least one concentration per year during March to June for at least four years during 1990 to 2005 was available, with the exception of Wisconsin where only three years of data were required. The less stringent data requirement for Wisconsin was used because there were insufficient data from Wisconsin to meet the four year requirement for inclusion.

The spatial pattern and magnitude of nitrate concentrations estimated from land use data (Figure 15a) are similar to those estimated from STORET data (Figure 15b). As might be expected, there is also reasonable concordance in spatial patterns of nitrate concentration, land use (Figure 15c), and likely extent of agricultural tile drainage (Figure 15d). Based on these comparisons, land use seems to provide a reasonable basis for estimating at least the general patterns of average stream nitrate concentrations across the UMR and Ohio River basins.

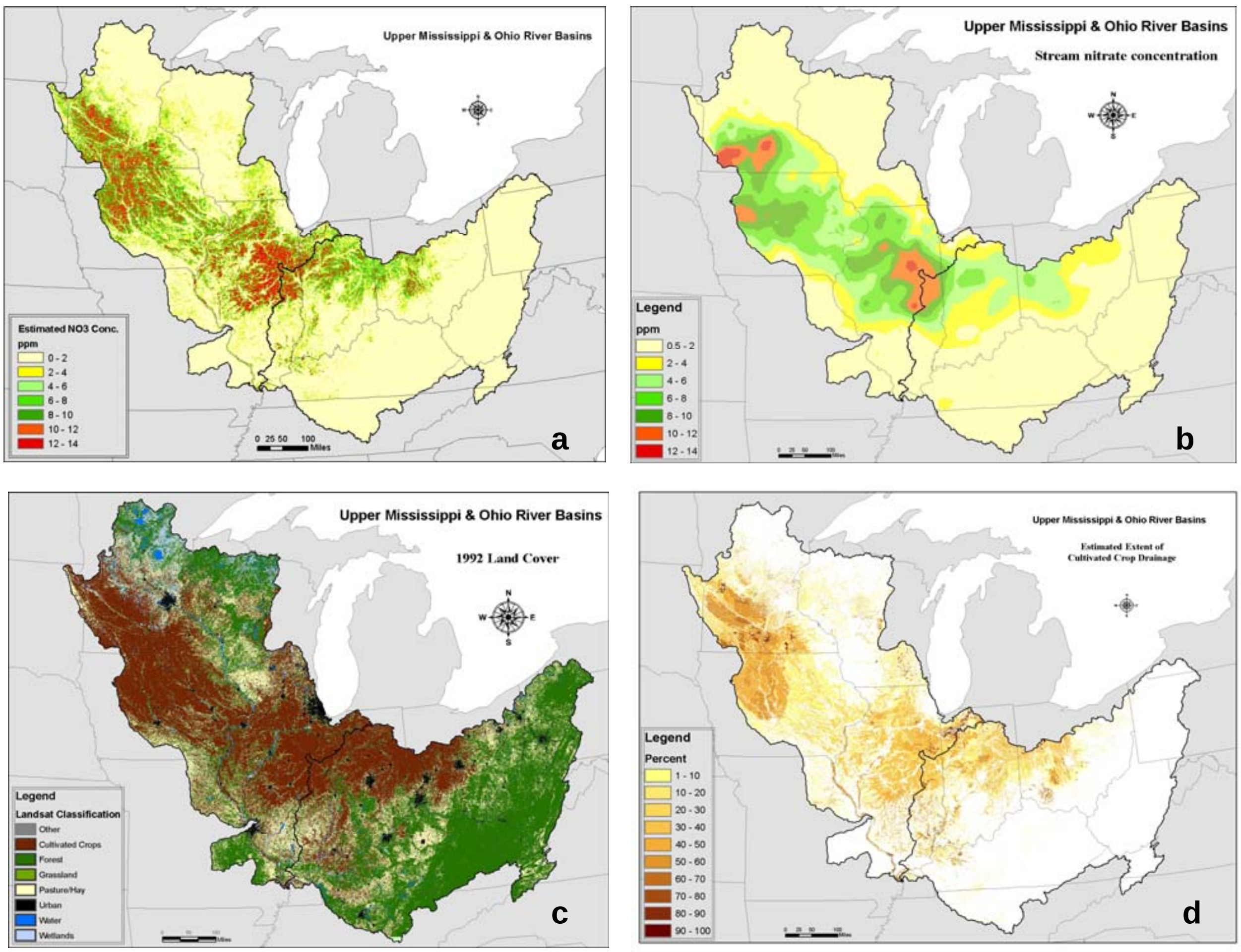

Figure 15. FWA nitrate concentrations estimated from land use (**a**) and STORET data (**b)**. Land cover based on Landsat data (**c**). Estimated extent of agricultural drainage based on distribution of corn and soybean on soils with a drainage class of poor or wetter, hydrologic group D, A/D,B/D, or C/D, and slopes 2% or less (**d**) (agricultural drainage estimated per D. Jaynes, National Soil Tilth Lab, Ames, IA).

GIS Based Estimates of Nitrate Load in the Upper Mississippi and Ohio River Basins

GIS based estimates of nitrate load were derived using a 100 ha grid covering the UMR and Ohio River basins. The annual water yield for each grid cell was estimated by interpolation of annual water yields from USGS stream monitoring stations with less than 1000 square mile watersheds and selected to encompass the UMR and Ohio River basins (Figure 16). Nitrate loading for each grid cell was calculated as the product of nitrate concentration (Figure 12a), water yield and grid cell area. The GIS based model discharge and nitrate loads for the basins were obtained by summation over the grid areas for each of the UMR and Ohio River basins. This procedure was repeated for each year in the simulation period (1990 to 1999).

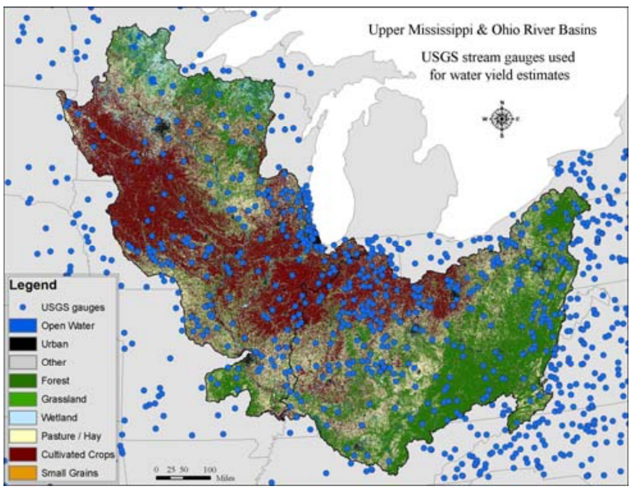

Figure 16. USGS stream monitoring stations used to estimate water yield.

The total annual discharge and nitrate load was estimated directly using the NASQAN data for comparison with the GIS model. The UMR discharge and nitrate loads were estimated by subtracting corresponding values at Hermann, MO on the Missouri River from values at Thebes, IL on the Mississippi River. The Ohio River basin discharge and nitrate loads were estimated by subtracting corresponding Tennessee River values from Ohio River values at Dam 53 near Grand Chain, IL. Tennessee River discharge and loads were estimated using data from the Tennessee River near Paducah, KY and at Savannah, TN. For years when data were missing for the station near Paducah, KY, discharge was estimated as the product of the discharge at Savannah, TN and the ratio of the watershed area at the Paducah to the watershed area at Savannah. Loads for missing years were determined as the product of the FWA nitrate concentration at Paducah (determined for years when data were available) and discharge. Using this approximation, the annual Tennessee River nitrate load varies between about 4% and 9.5% of the annual Ohio River nitrate load from 1980 to 2005. The discharges and loads determined directly from USGS flow data and NASQAN nitrate data show a generally good agreement with the GIS model based loads for these river systems (Figure 17).

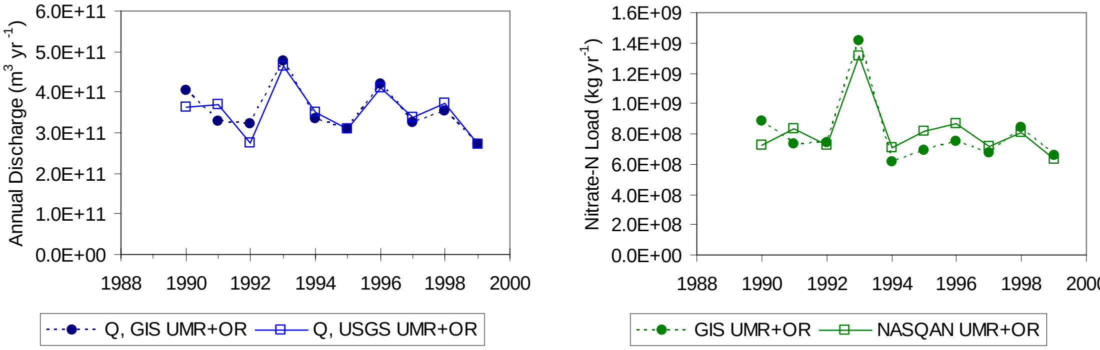

Figure 17. GIS water yield grid and nitrate-N load grid sums show good agreement with USGS annual discharge (R^2 = 0.79) and NASQAN nitrate-N load (R^2 = 0.74).

GIS Based Estimates of Potential Nitrate Reduction by Wetlands in the Upper Mississippi and Ohio River Basins

Wetland nitrate removal performance was estimated for fifty-four hypothetical wetland restoration sites distributed across the UMR and Ohio River basins (Figure 18). Separate mass balance models were created for each site and simulations were conducted for drainage basins with wetland pools providing wetland/watershed area ratios of 1-4%. Simulations were run using a daily time step over the ten year period from 1990 to 1999. Each model was run for one year prior to 1990 to remove effects associated with initial model conditions. Nitrate removal was modeled as a temperature-dependent, first-order process for loss rate coefficients from 25-75 m yr^{-1} (Crumpton 2001). Daily hydrologic inflows were estimated from local water yield based on USGS gauging station data. Temperature data were collected from the National Oceanic and Atmospheric Administration (NOAA) NNDC Climate Data Online website. Several thousand weather stations across the region were screened for the most complete data sets. The NOAA weather stations in closest proximity to each simulation point were used for temperature inputs. Where intervals of temperature data were missing, substitute values from nearby stations were used to provide a complete daily temperature record for the ten year period for each of the 54 simulation sites. The average of the observed minimum and maximum daily temperature was used as the daily temperature input for modeling. Input temperature data were conditioned to have a minimum value of 4˚C, which is the approximate temperature of liquid water in wetland sediment during cold winter months. Daily nitrate mass inflow was calculated as the product of estimated nitrate concentration and daily hydrologic inflow. Outflow was controlled by a weir equation. Daily nitrate mass outflow from the wetland was the product of simulated wetland output nitrate concentration and daily hydrologic outflow.

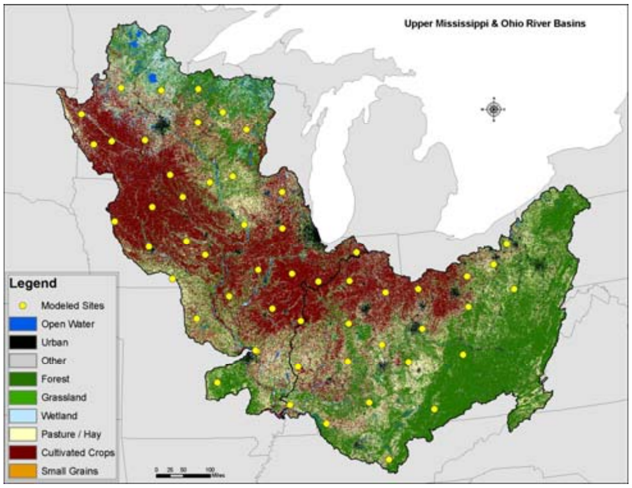

Figure 18. Wetland simulation sites.

As in the previous analyses (Figures 6 and 7), percent nitrate removal in wetland simulations is largely a function of annual hydraulic loading rate (Figure 19) and to a lesser extent a function of temperature. Also as in the previous analysis, mass loss can be predicted as the product of percent loss and mass load (Figure 19).

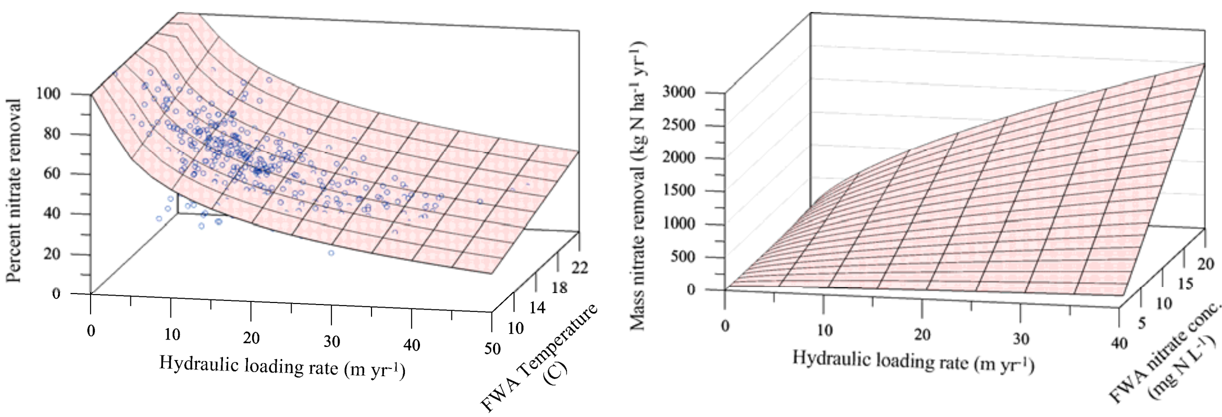

Figure 19. Left panel: percent nitrate removal declines with increasing hydraulic loading rate (HLR) to the wetland and increases slightly with average water temperature weighted to account for timing of load. Variability reflects the combined effects of annual variability in timing of hydraulic loading and temperature. $R^2 = 0.748$, n=540 (54 sites by 10 years). Right panel: expected mass nitrate removal increases with increasing hydraulic loading and nitrate concentration.

GIS based estimates of potential nitrate removal by wetlands were derived for the same grid used for load estimates in the UMR and Ohio River basins. The potential mass reduction of nitrate by wetlands was estimated on the basis of the expected nitrate load and the percent removal expected (primarily a function of annual HLR). The annual water yield for each grid cell was estimated by interpolation of annual water yields from USGS stream monitoring stations with less than 1000 square mile watersheds and selected to encompass the UMR and Ohio River basins (Figure 16). Nitrate concentration (Figure 15a) for each grid cell was estimated based on % RC grid using the regression developed for NASQAN sites (Figure 11). Nitrate loading to wetlands was calculated as the product of nitrate concentration, water yield, and drainage area above the wetland. The HLR to wetlands was calculated based on the water yield, wetland area, and drainage area above the wetland. Percent nitrate removal was estimated based on HLR functions (Figure 19) spanning a 3 fold range in loss rate coefficient (Crumpton 2001) and encompassing the observed performance reported for wetlands in the UMR and Ohio River basins (Table 2, Figure 7). The nitrate load was multiplied by the expected percent nitrate removal to estimate the mass removal. This procedure was repeated for each restoration scenario each year in the simulation period (1990 to 1999).

The spatial distribution of nitrate mass loading across the UMR and Ohio River basins is shown in Figure 20 as the 1990s average mass load in kg nitrate-N km^{-2} of watershed $year^{-1}$. As could be expected, mass loads are greatest in those areas with extensive row crop (compare Figure 15c). On the basis of the GIS modeling results, less than 30% of the UMR and Ohio River basins contributes 80% of their combined nitrate load to the Mississippi River. The spatial distribution of nitrate mass removal by wetlands is shown in Figure 21 as the 1990s average removal rate in kg nitrate-N ha^{-1} of wetland $year^{-1}$ for a scenario with a wetland/watershed area ratio of 2%. The spatial pattern in mass removal (Figure 21) was similar for all restoration scenarios; with of course lower removal rates for more conservative scenarios and higher removal rates for less conservative scenarios. For a wetland/watershed area ratio of 2%, GIS modeling results indicate that a 30% reduction in the total nitrate load exported from the UMR and Ohio River basins could be achieved with approximately 210,000 to 450,000 ha of restored wetlands (Figure 22), assuming the wetlands could be located so as to intercept water from the highest nitrate load contributing areas. The wetlands would be expected to remove on average 40-60 % of the load received, have a cumulative average mass reduction of 830 kg nitrate-N ha^{-1} of wetland $year^{-1}$, and have a range in cumulative average mass reduction of 530-1130 kg nitrate-N ha^{-1} of wetland $year^{-1}$.

These results are based on the assumption that the FWA nitrate concentration versus percent row crop regression provides a reasonable estimate of FWA nitrate concentration. Comparison of GIS model output with observed loads (Figure 17) indicates that this approach reasonably predicts loads at large scale. However, the rivers on which the land use regression is based drain >200 square mile watersheds. Even large tile drainage networks typically drain areas of less than a few square miles and nitrate concentrations in tile drainage water can be significantly greater than concentrations estimated here based on the land use regression for NASQAN watersheds (Baker et al. 1997; 2004; in press; David et al. 1997; Sawyer and Randall in press). Wetlands could achieve significantly greater nitrate reductions than estimated here if nitrate concentrations are significantly higher than assumed in these analyses. However, the results also assume wetlands will be restored in areas of extensive row crop where nitrate mass loads are greatest. If wetlands are instead restored in areas with lower nitrate concentrations and loads, then the wetlands would be expected to remove significantly less nitrate than estimated here.

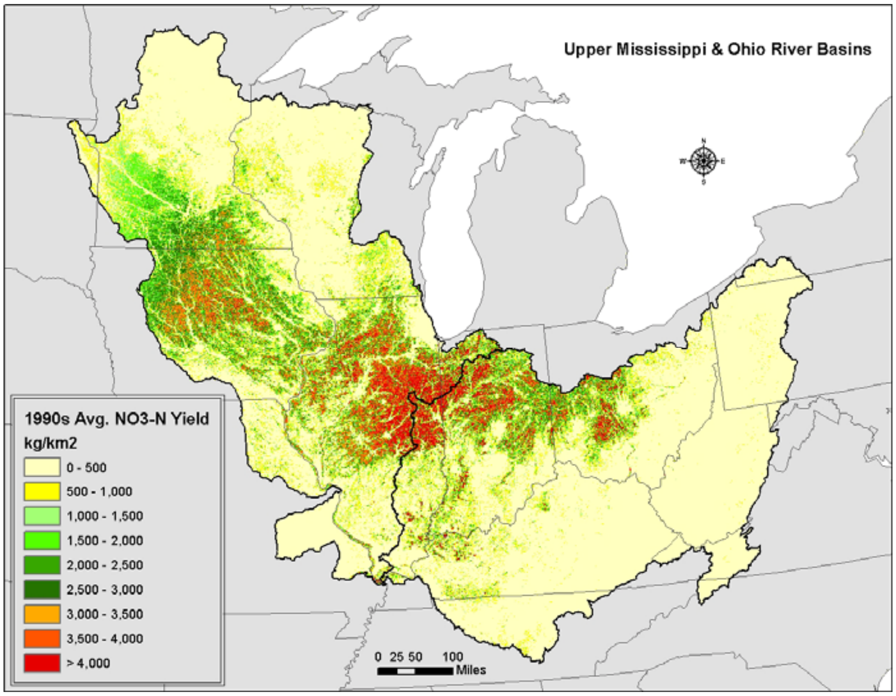

Figure 20. Estimated average nitrate load in kg N/km^2 of watershed/year.

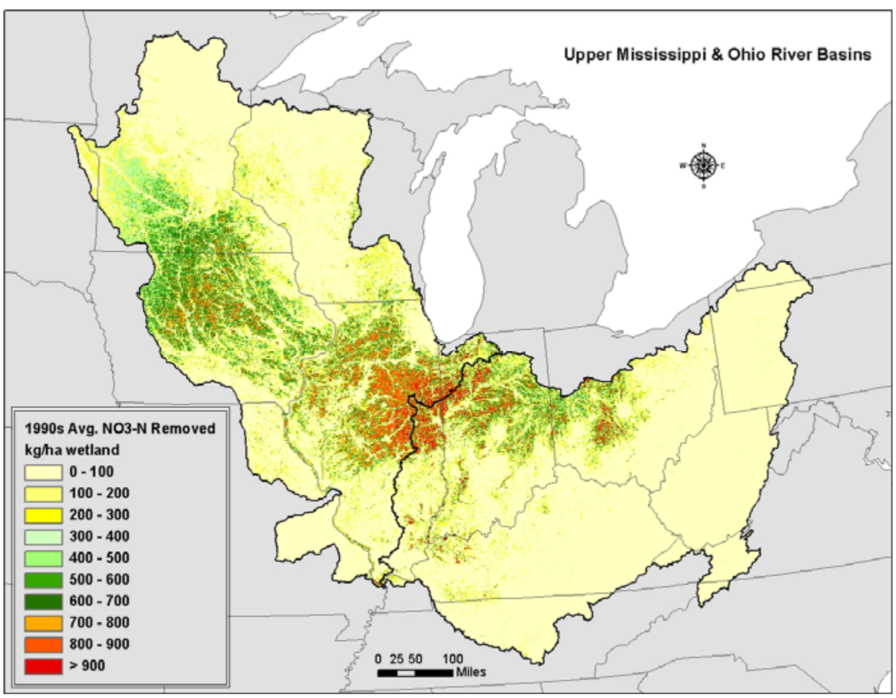

Figure 21. Estimated average nitrate removal in kg N/ha of wetland /year for wetlands with a 2% wetland/watershed ratio.

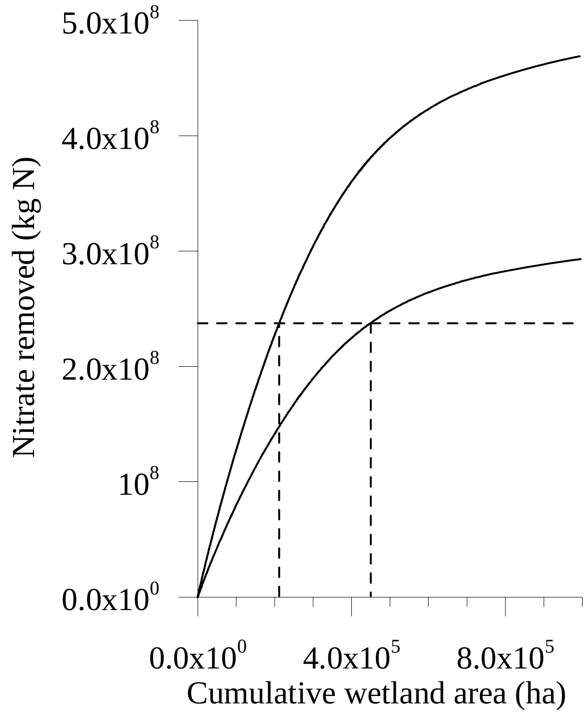

Figure 22. Modeled range of cumulative nitrate reduction in the Upper Mississippi River and Ohio River basins for targeted wetland restorations with a wetland/watershed ratio of 2%. The upper and lower curves represent the least conservative and most conservative scenarios modeled with respect to wetland denitrification capacity. The horizontal dashed line represents 30% of the 1990s average mass load from the UMR and Ohio River basins. The vertical dashed lines indicate the range of wetland areas needed to remove a mass of nitrate equivalent to 30% of the 1990s average annual load from the basins.

The long term patterns in nitrate loading and removal summarized here obscure relatively large annual and geographic variation in nitrate loading. The annual patterns in the spatial distribution of water yield, nitrate loads, and nitrate removal by wetlands with a 2% wetland/watershed area ratio are illustrated in figures 23 to 32 for 1990 through 1999. These figures cover a range of wet to dry years within each sub-basin, although wet or dry years in the UMR basin do not necessarily correspond to wet or dry years, respectively, in the Ohio River basin. For example, 1993 (Figure 26) was a flood year for much of the UMR basin, but was a fairly typical year for the Ohio River basin, and yet the nitrate load at the confluence of these rivers was greater than any other year during 1990 to 1999. In contrast, 1994 (Figure 27) had relatively low flow in the UMR basin, with fairly heavy flow concentrated in the southern and eastern portions of the Ohio River basin, and this year had relatively low nitrate load at the confluence of these rivers.

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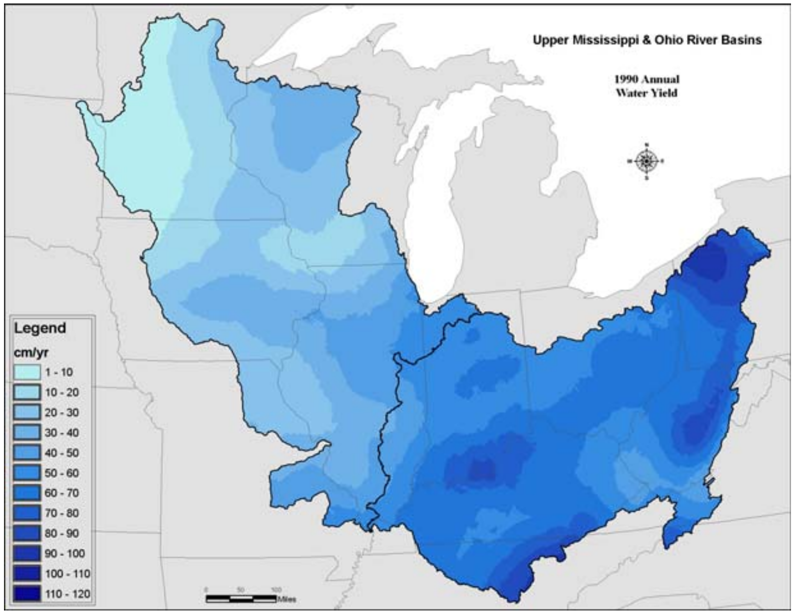
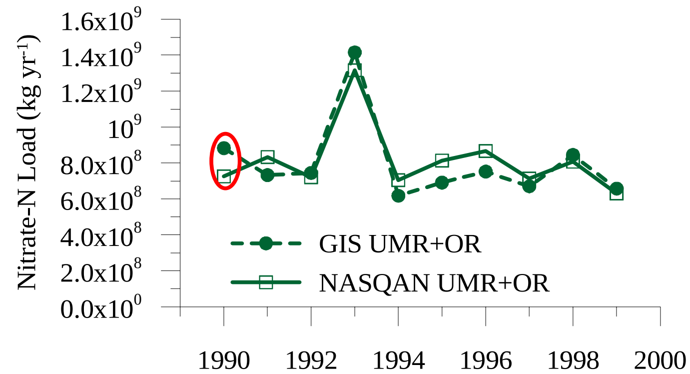
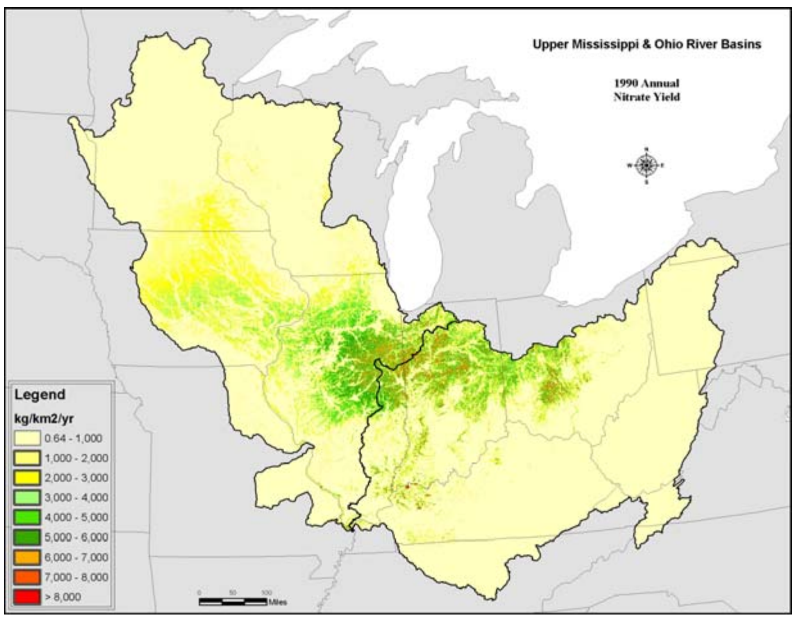
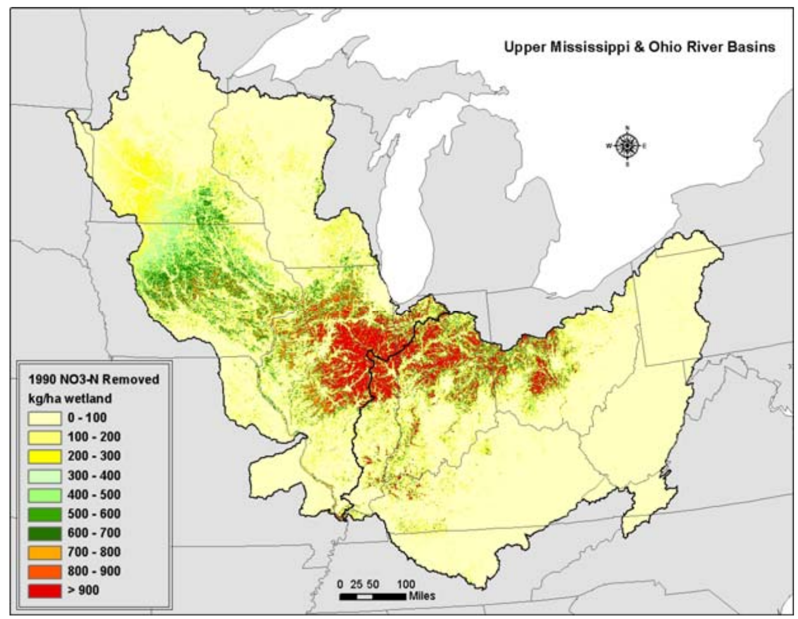


Figure 23. 1990 water yield, nitrate yield, and modeled nitrate loss for wetlands with a 2% wetland/watershed ratio.







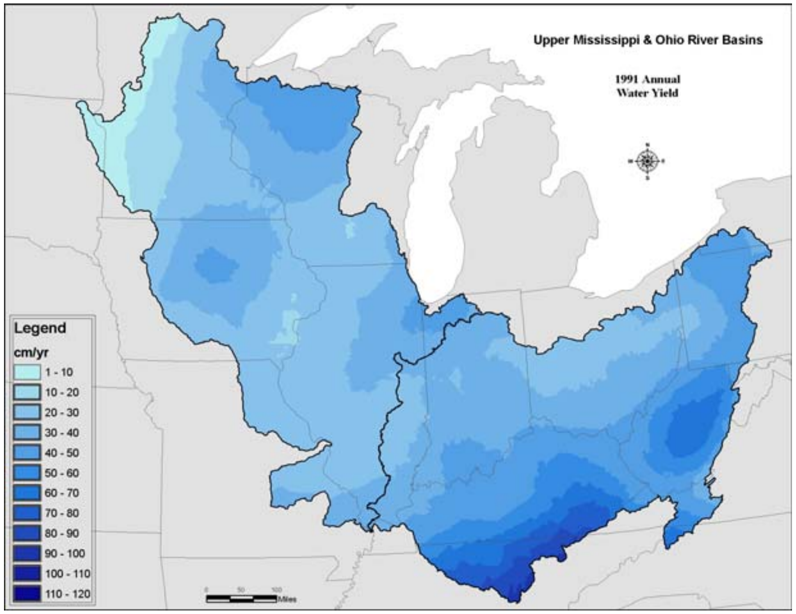
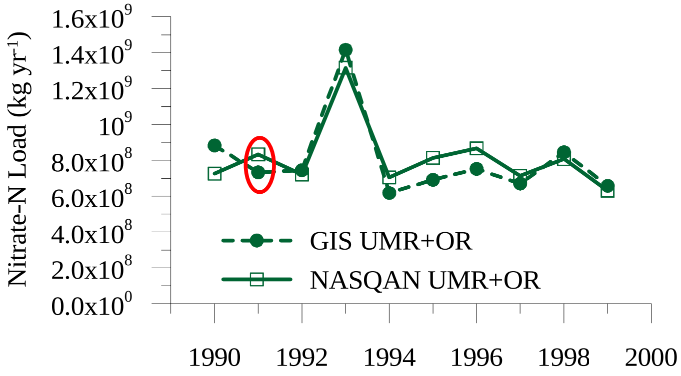
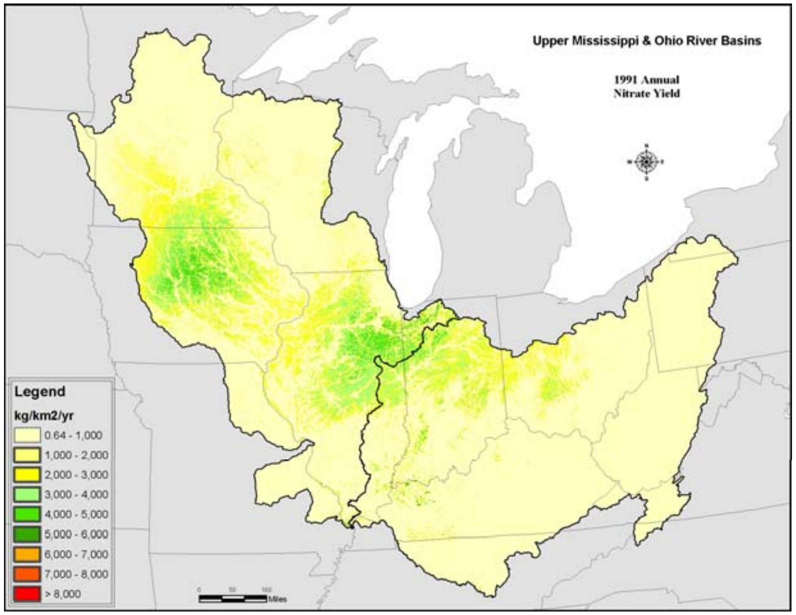
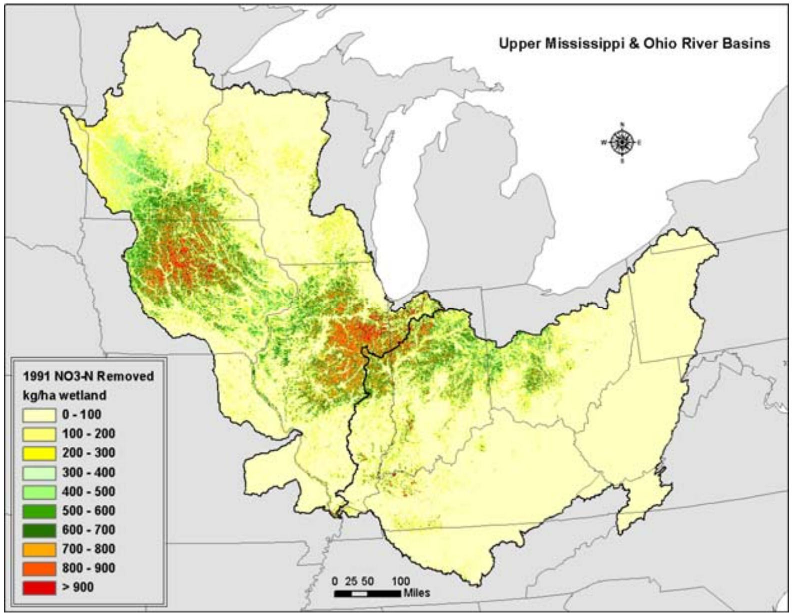


Figure 24. 1991 water yield, nitrate yield, and modeled nitrate loss for wetlands with a 2% wetland/watershed ratio.







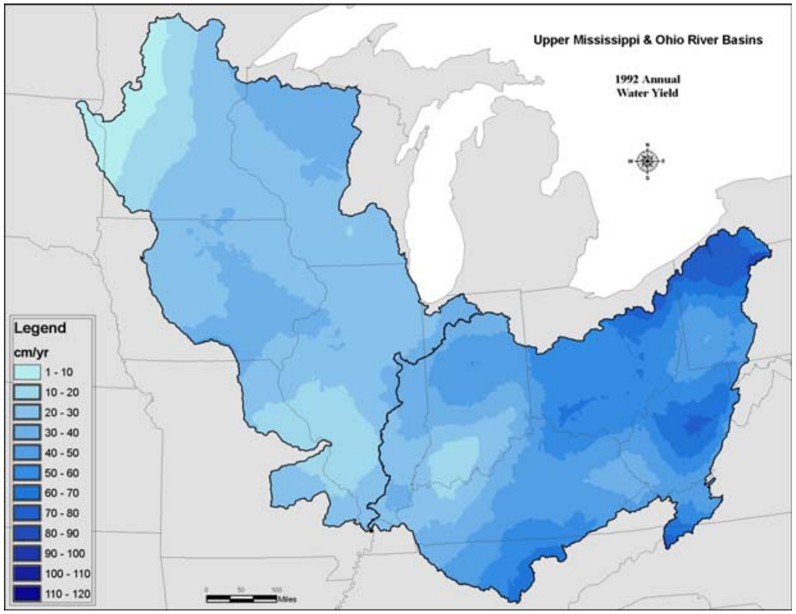
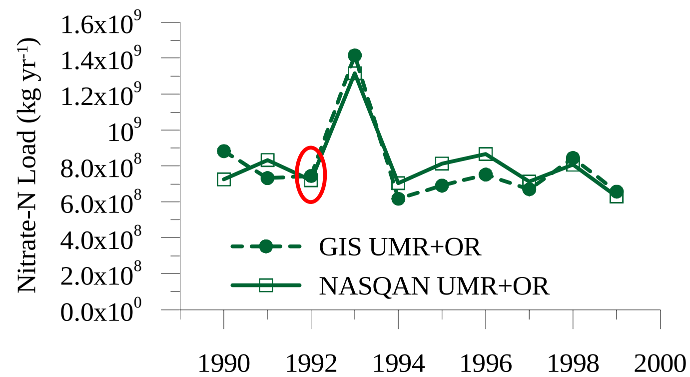
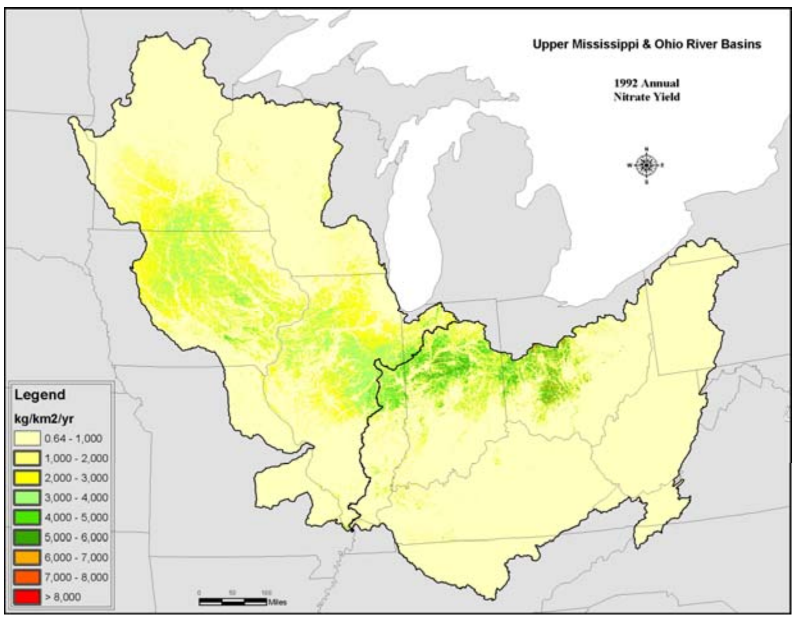
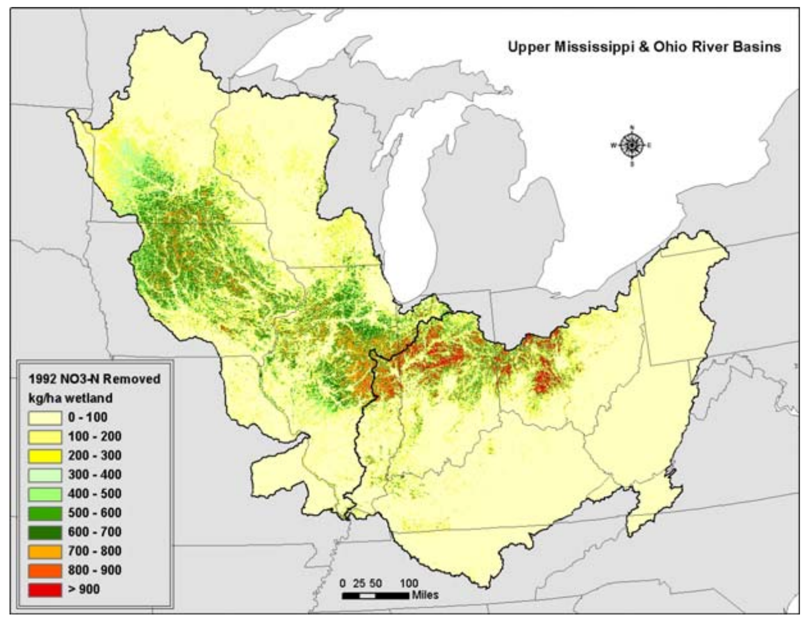


Figure 25. 1992 water yield, nitrate yield, and modeled nitrate loss for wetlands with a 2% wetland/watershed ratio.







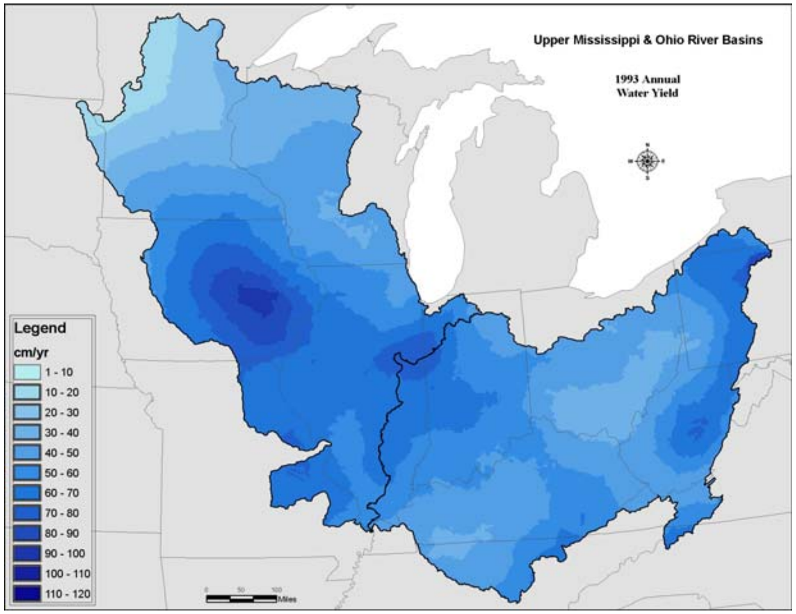
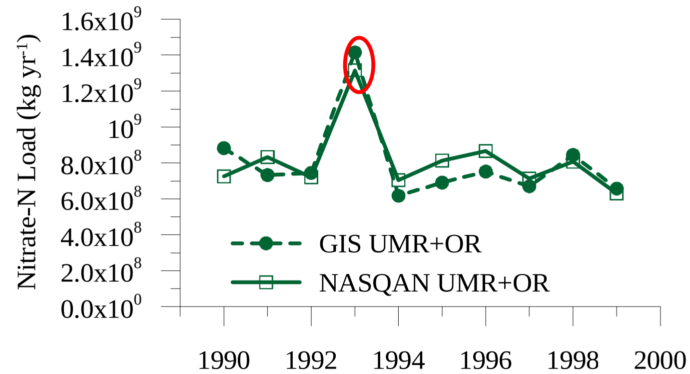
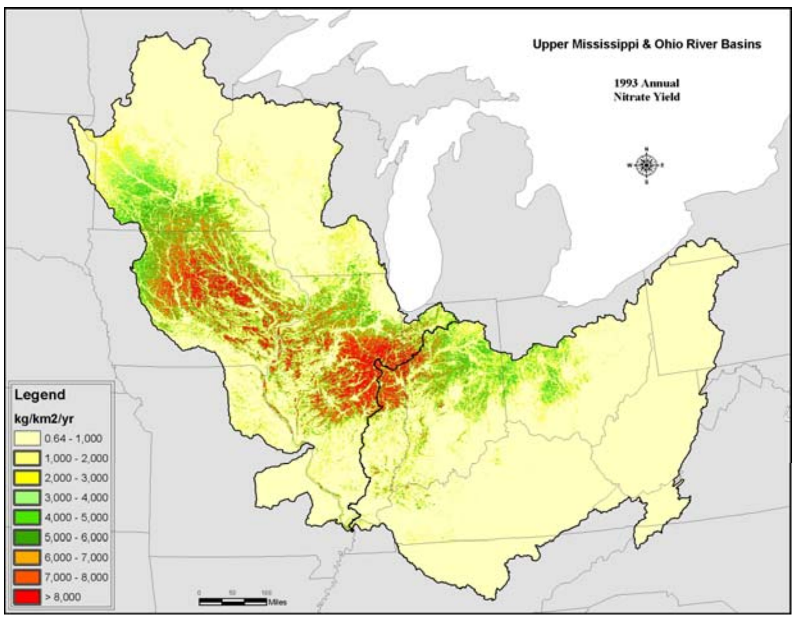
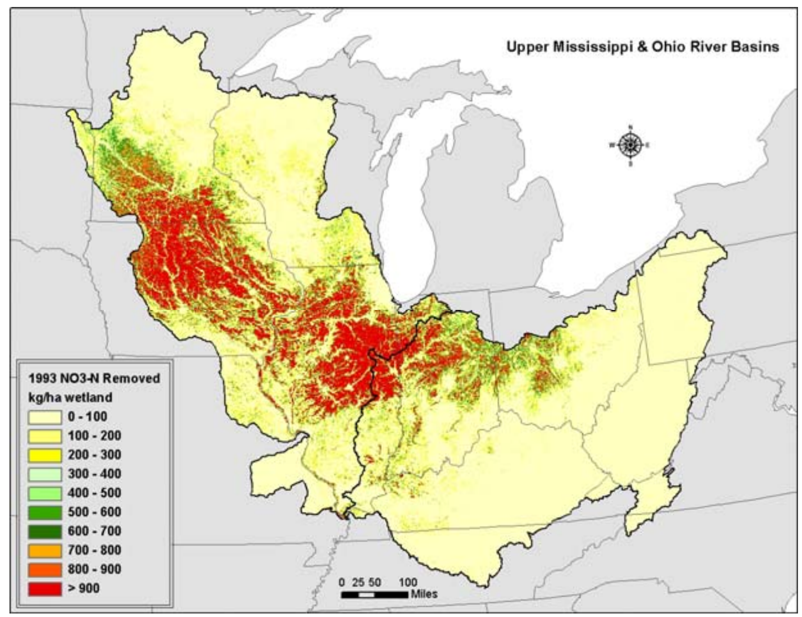


Figure 26. 1993 water yield, nitrate yield, and modeled nitrate loss for wetlands with a 2% wetland/watershed ratio.







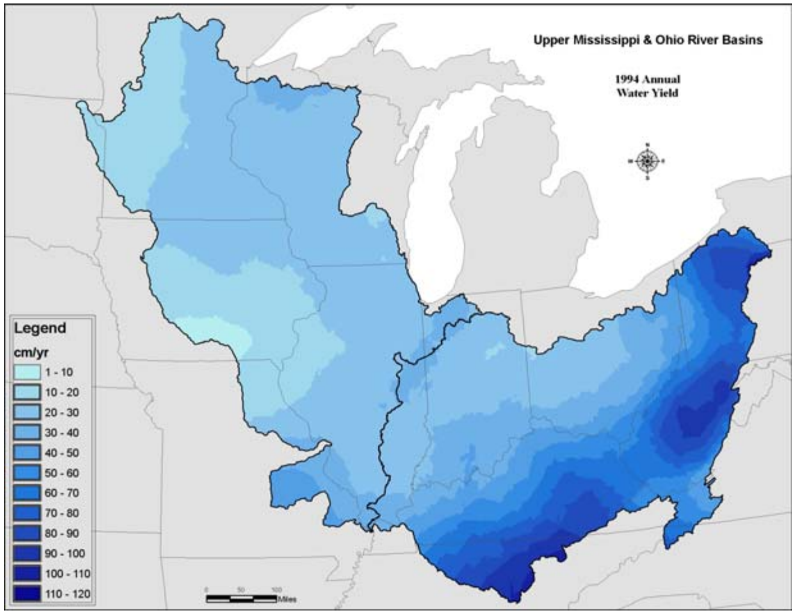
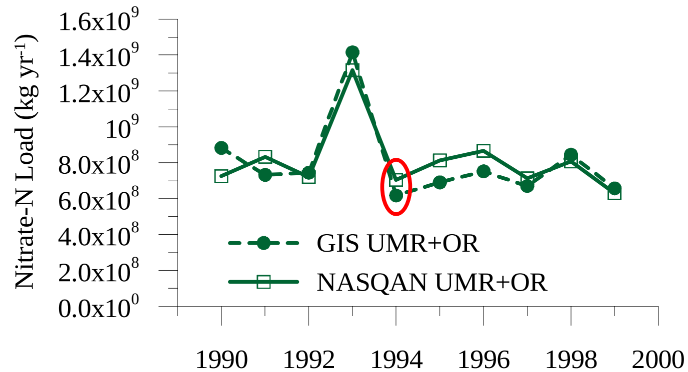
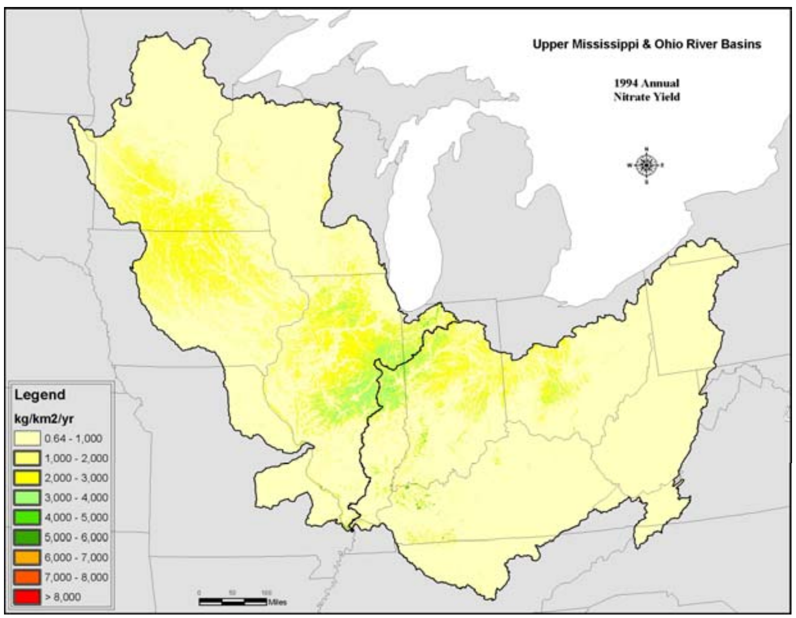


Figure 27. 1994 water yield, nitrate yield, and modeled nitrate loss for wetlands with a 2% wetland/watershed ratio.





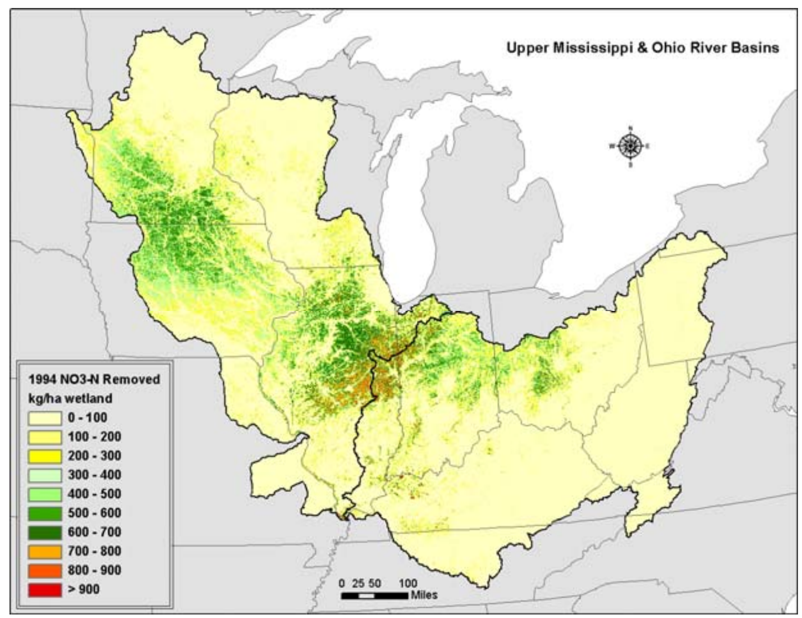
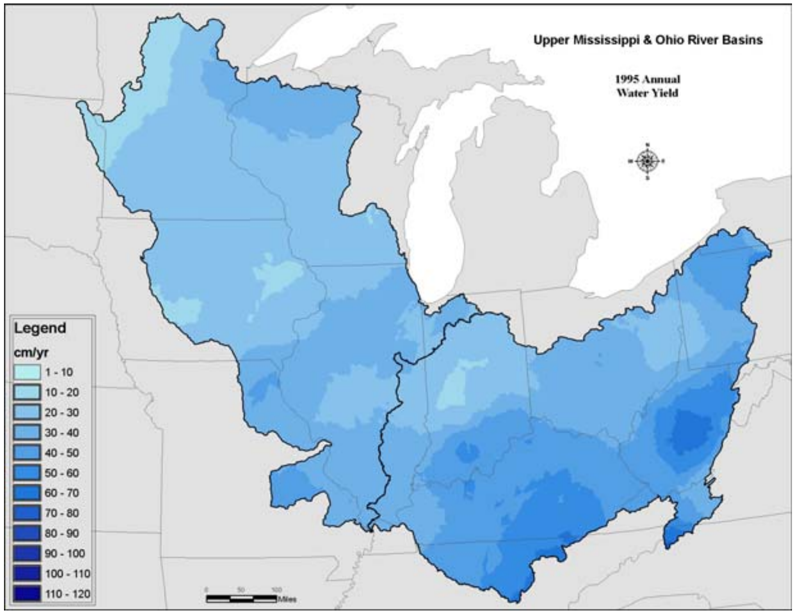
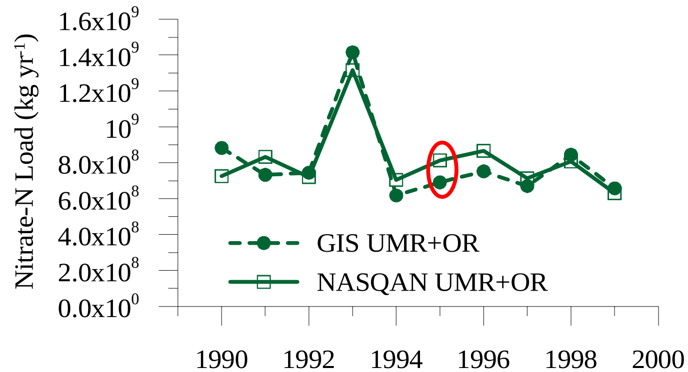
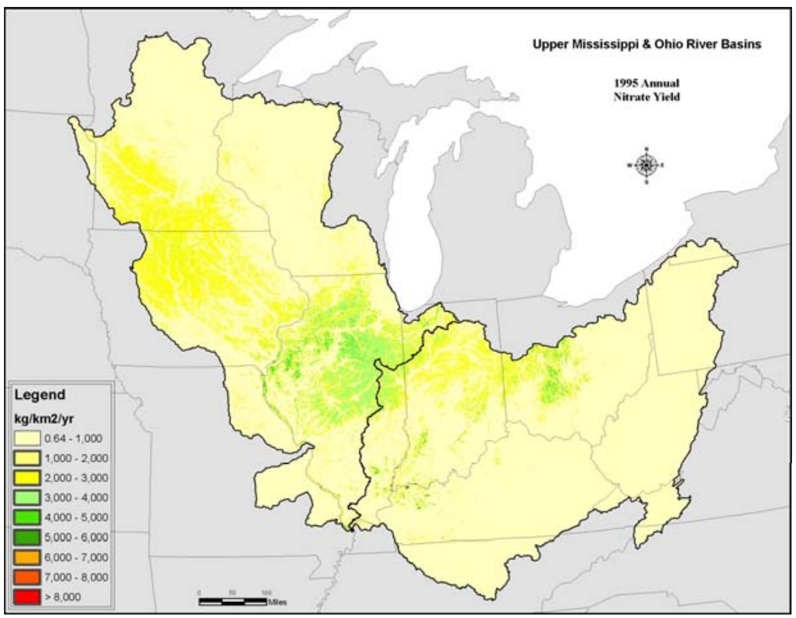
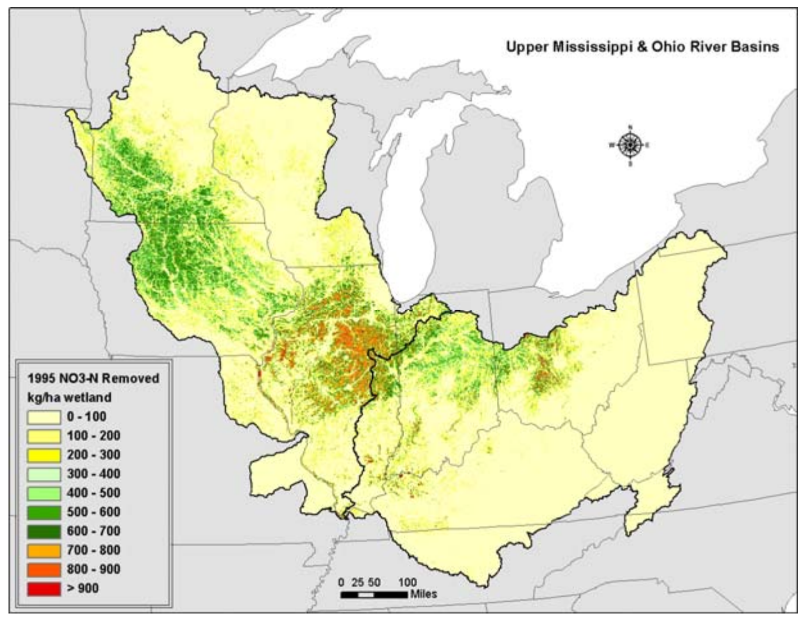


Figure 28. 1994 water yield, nitrate yield, and modeled nitrate loss for wetlands with a 2% wetland/watershed ratio.









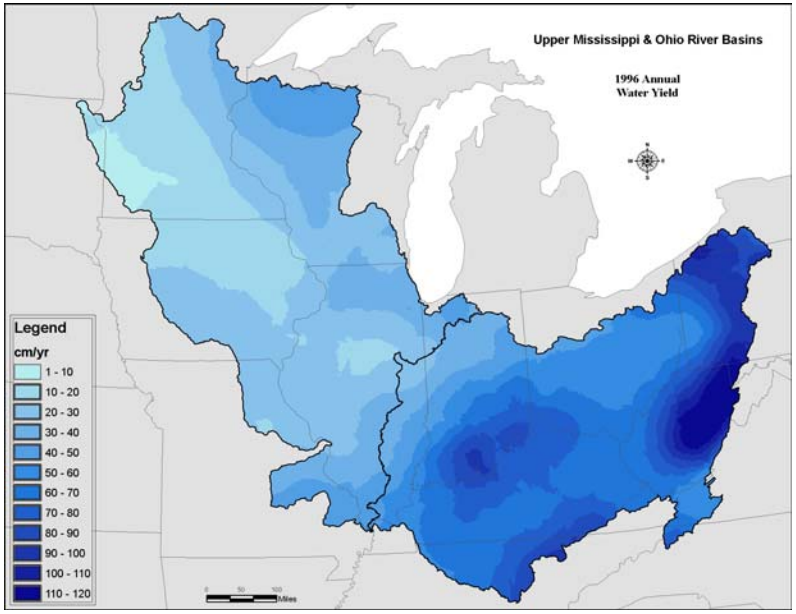
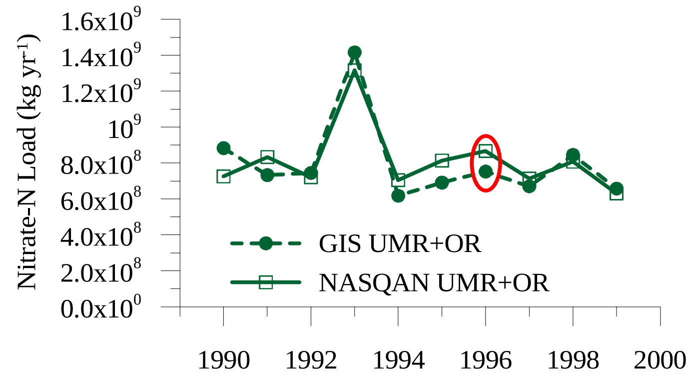
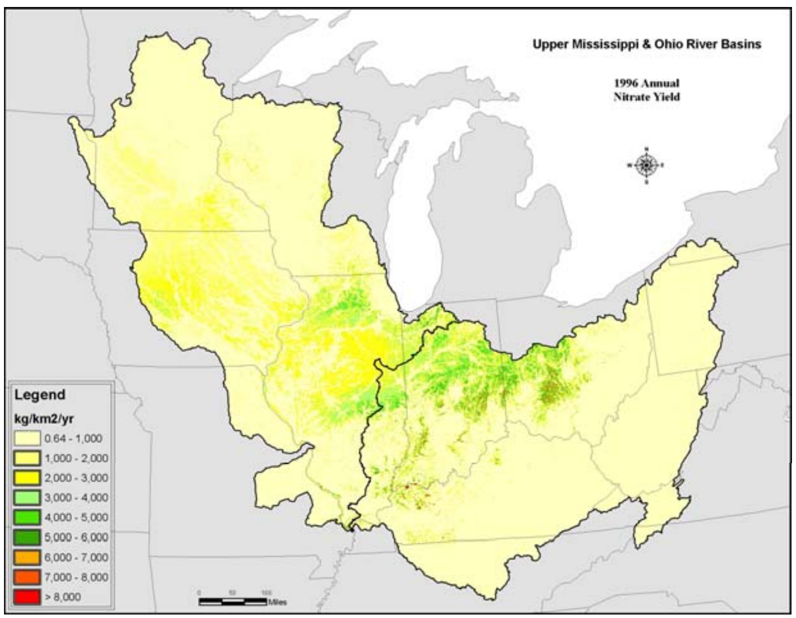


Figure 29. 1996 water yield, nitrate yield, and modeled nitrate loss for wetlands with a 2% wetland/watershed ratio.





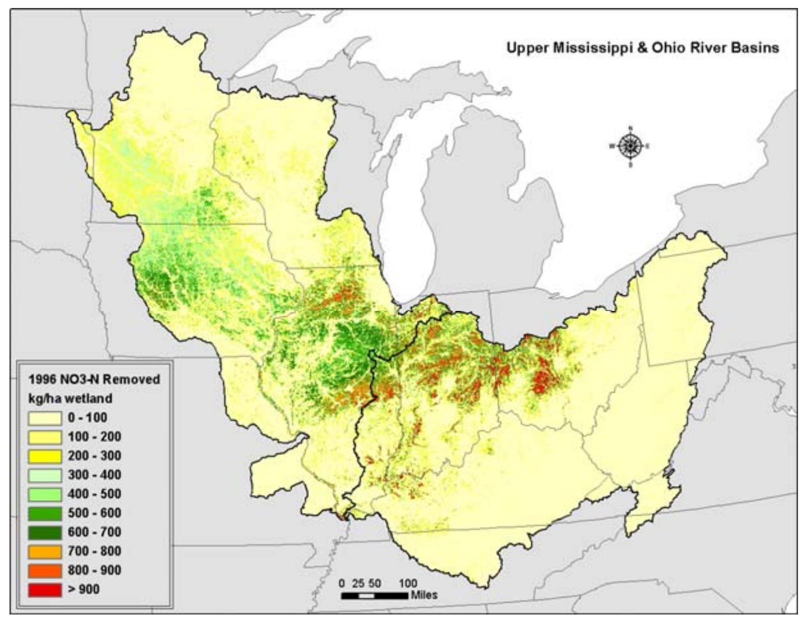
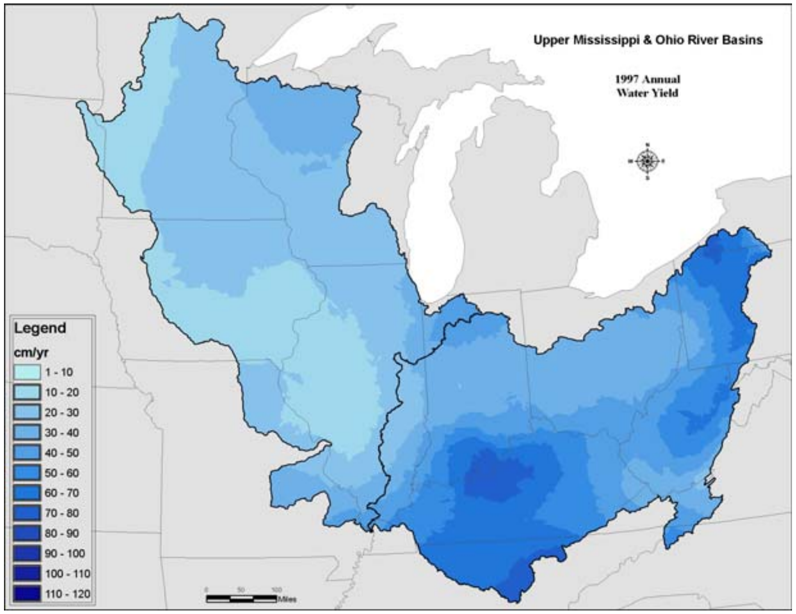
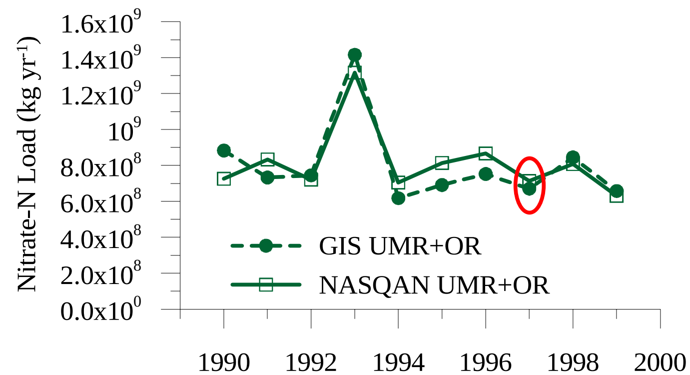
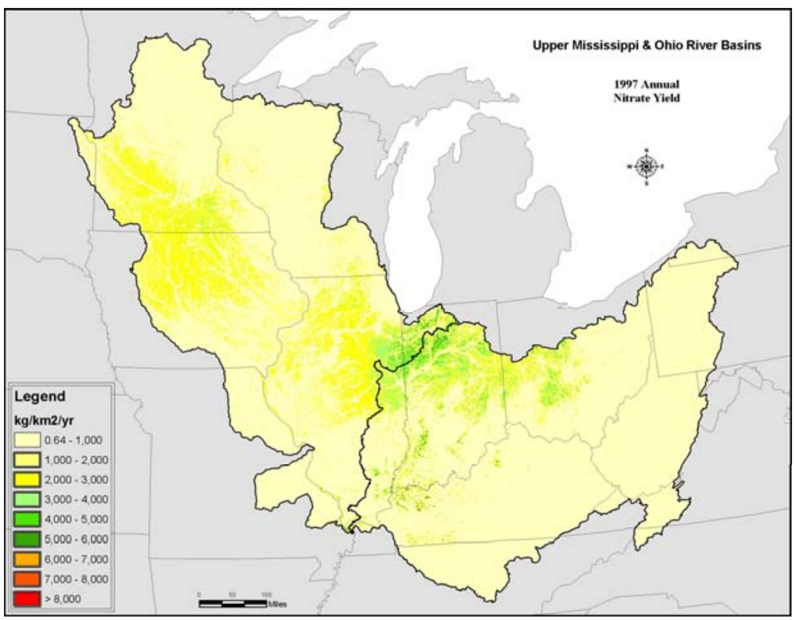
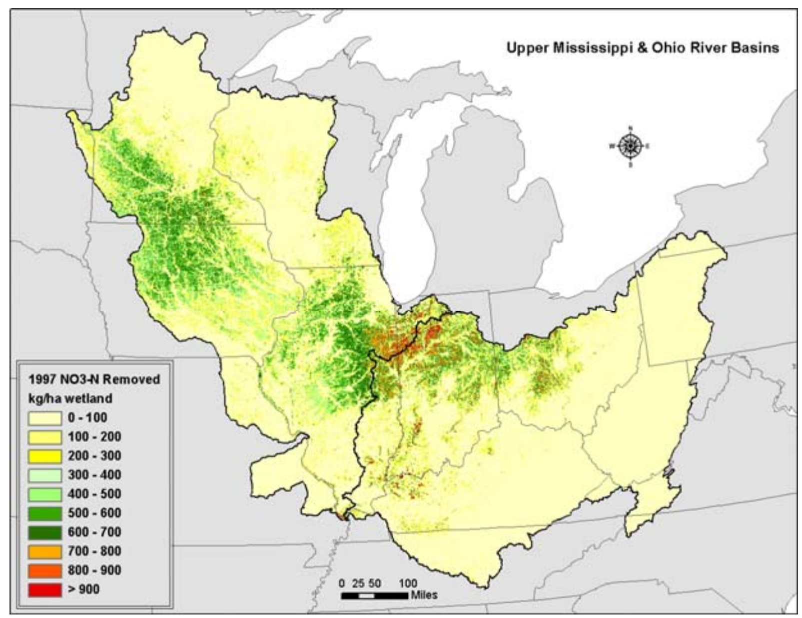


Figure 30. 1997 water yield, nitrate yield, and modeled nitrate loss for wetlands with a 2% wetland/watershed ratio.









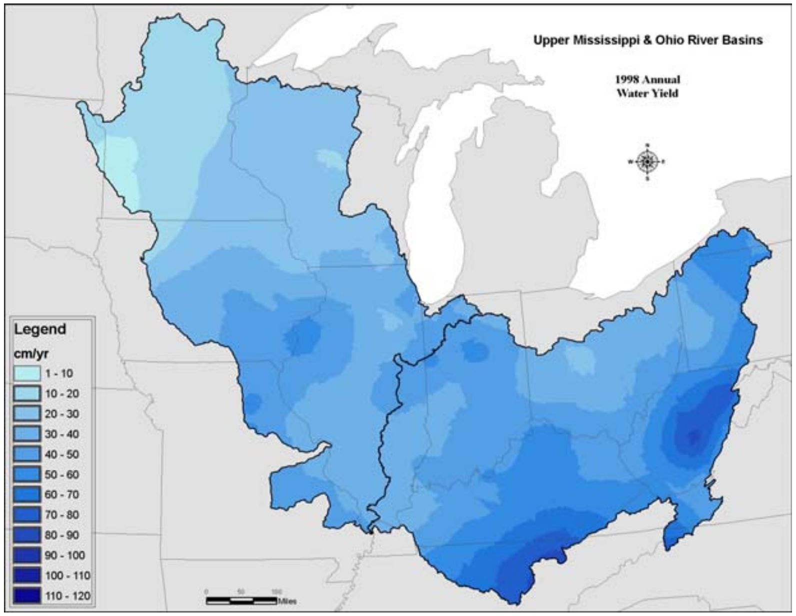
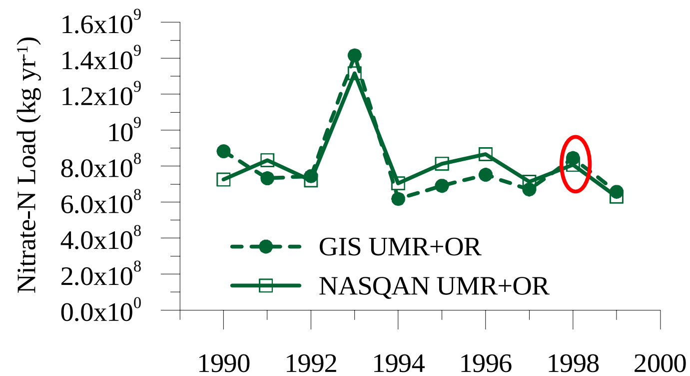
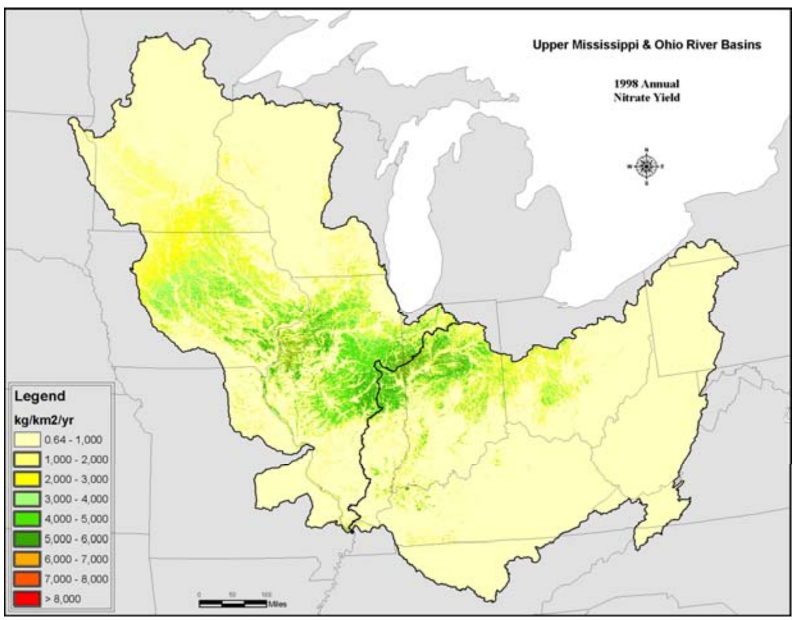
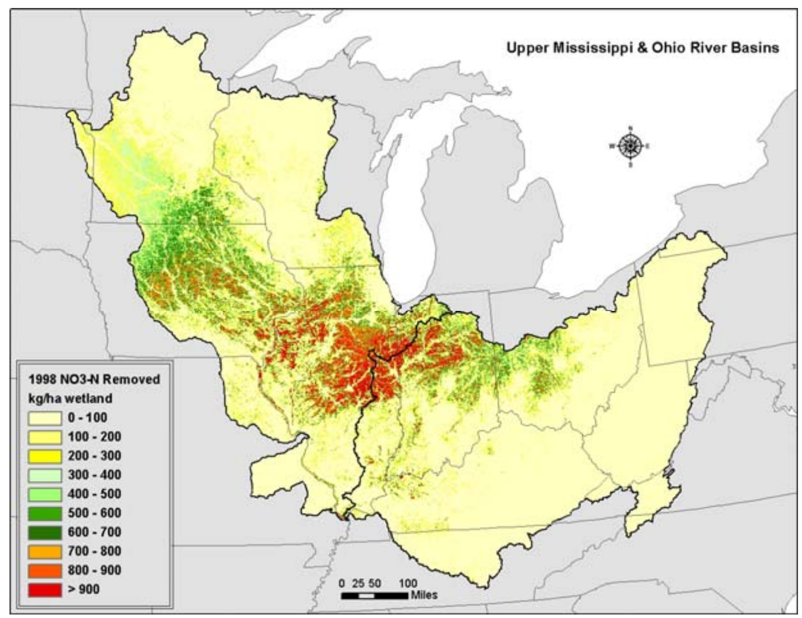


Figure 31. 1998 water yield, nitrate yield, and modeled nitrate loss for wetlands with a 2% wetland/watershed ratio.







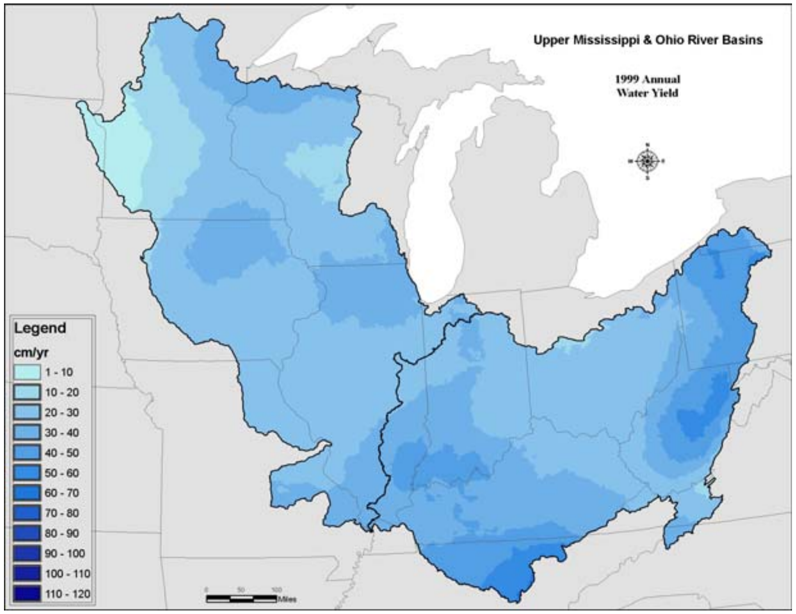


Figure 32. 1999 water yield, nitrate yield, and modeled nitrate loss for wetlands with a 2% wetland/watershed ratio.