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Toward disentangling the effect of hydrologic and nitrogen source changes from 1992 to 2001 on incremental nitrogen yield in the contiguous United States

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The goal of this research was to quantify the relative impact of hydrologic and nitrogen source changes on incremental nitrogen yield in the contiguous United States. Using nitrogen source estimates from various federal data bases, remotely-sensed land use data from the National Land Cover Data (NLCD) program, and observed instream loadings from the United States Geological Survey (USGS) National Stream Quality Accounting Network (NASQAN) program, we calibrated and applied the spatially-referenced regression model SPARROW to estimate incremental nitrogen yield for the contiguous United States. We ran different model scenarios to separate the effects of changes in source contributions from hydrologic changes for the years 1992 and 2001, assuming that only state conditions changed and that model coefficients describing the stream water-quality response to changes in state conditions remained constant between 1992 and 2001. Model results show a decrease of 8.2% in the median incremental nitrogen yield over the period of analysis with the vast majority of this decrease due to changes in hydrologic conditions rather than decreases in nitrogen sources. For example, when we changed the 1992 version of the model to have nitrogen source data from 2001, the model results showed only a small increase in median incremental nitrogen yield (0.12%). However, when we changed the 1992 version of the model to have hydrologic conditions from 2001, model results showed a decrease of approximately 8.7% in median incremental nitrogen yield. We did, however, find notable differences in incremental yield estimates for different sources of nitrogen after controlling for hydrologic changes, particularly for population related sources. For example, the median incremental yield for population related sources increased by 8.4% after controlling for hydrologic changes. This is in contrast to a 2.8% decrease in population related sources when hydrologic changes are included in the analysis. Likewise we found that median incremental yield from urban watersheds increased by 6.8% after controlling for hydrologic changes – in contrast to the median incremental nitrogen yield from cropland watersheds, which decreased by 2.1% over the same time period. These results suggest that, after accounting for hydrologic changes, population related sources became a more significant contributor of nitrogen yield to streams in the contiguous United States over the period of analysis. However, this study was not able to account for the influence of human management practices such as improvements in wastewater treatment plants or Best Management Practices (BMPs) that likely improved water quality, due to a lack of data for quantifying the impact of these practices for the study area.

Keywords: Water quality, anthropogenic activities, land use, non-linear regression modeling, hydrology.

1. Introduction

Anthropogenic activities are altering the nitrogen-cycle and resulting in increased contributions of excess nitrogen to aquatic ecosystems [*Howarth et al.*, 1996; *Galloway et al.*, 1995; *Boyer et al.*, 2002]. This excess nitrogen can have serious environmental impacts including algal blooms that contribute to anoxic and hypoxic conditions in waterbodies

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[*NRC*, 2000]. The frequency and magnitude of hypoxic areas in coastal waterbodies in the United States and abroad have increased during the latter half of the twentieth century [*Diaz*, 2001; *Rabalais et al.*, 2002]. In the Gulf of Mexico in particular, increased nitrogen loading from the Mississippi River has caused eutrophication and chronic seasonal hypoxia in the shallow waters of the Louisiana Shelf [*Alexander et al.*, 2000]. Many large ecosystems that are now severely stressed by hypoxia face declines in fishery production, reductions in species diversity, and changes to food web structures [*Diaz*, 2001]. Studies suggest that future riverine nitrogen export is likely to increase by as much as 24% in response to heavier fertilizer use, expanded corn production to meet the increased demand of food and bio-fuel production, and an increase in annual river discharge under future climate conditions [*Han et al.*, 2009]. These factors make it important to understand how regional-scale anthropogenic activities, in addition to direct changes to nitrogen sources and hydrology, will impact the delivery of nitrogen to waterbodies.

Nitrogen sources and delivery involve interrelationships between human, economic, and physical systems. Anthropogenic activities are not only altering the spatial distribution of nitrogen sources, but also the hydrologic cycle by changing land use, which in turn changes

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evapotranspiration rates, precipitation rates, runoff volumes, infiltration rates, and air temperature [*Feddema et al.*, 2005; *Foley et al.*, 2005; *Juang et al.*, 2007]. Application of nitrogen, increased in the form of fertilizer, is driven by food, and more recently bio-fuel, production needs [*Stonestrom et al.*, 2009]. Human activities are also responsible for the change of animal waste input and atmospheric deposition. Nitrogen delivery is dependent on a watershed's ecological and biophysical characteristics including soil properties, stream availability, average annual temperature and precipitation [*Smith et al.*, 1997; *Jones et al.*, 2001]. For these reasons, quantifying the impacts of anthropogenic nitrogen sources and hydrology/climate related changes in the contiguous United States, the main focus of this study, remains a major research challange.

Past data collection efforts in the United States have resulted in spatially detailed information on nitrogen sources, the physical environment, and instream nitrogen concentrations. Modelers have leveraged these historical data to understand the relationships that drive nitrogen fate and transport at regional spatial scales. The SPAtially Referenced Regressions On Watershed Attributes (SPARROW) model [*Smith et al.*, 1997] is an example of a regional-scale nitrogen fate and transport model that uses a combination of physically-based process representations (e.g., mass balances, overland and instream losses) and statistical regression to explain observed spatial variability in instream nitrogen concentrations. SPARROW has been applied to the Mississippi River Basin [*Alexander and Smith*, 2005; *Alexander et al.*, 2008], the Chesapeake Bay Watershed [*Roberts and Prince*, 2010], the contiguous United States [*Smith et al.*, 1997], and other regions in the United States and abroad [*Elliott et al.*, 2005; *Hoos and McMahon*, 2009]. SPARROW is unique as a statistical water quality model because it incorporates spatial referencing of the watershed and river network when modeling the transfer of nutrients from the landscape to streams and through the river network. *Smith et al.* [1997] demonstrated that spatial referencing increases model accuracy by reducing commonly encountered problems of network sparseness, bias, and basin heterogeneity. Although the model has been widely used for regional scale analysis, only one study has used the model to understand temporal changes [*Alexander et al.*, 2008], and this study was limited to the Mississippi River Basin.

The objective of this study was to quantify the relative effects of changes in climate/hydrology compared to changes in nitrogen source contributions to incremental nitrogen yield for the contiguous United States over a decadal time period (1992-2001). We applied SPARROW to estimate changes in the delivery of nitrogen to waterbodies for the years 1992 and 2001 due to changes in source contributions and hydrological factors. We used these two years because of the availability of National Land Cover Dataset (NLCD) that includes a land use change product dataset for 1992 and 2001. Using datasets described in the following section, we parameterized and then calibrated SPARROW for the contiguous United States using the best available information for the year 1992. We then used the calibrated model coefficients to predict incremental nitrogen yield for 1992, 2001, and two other hypothetical scenarios: one where we set the hydrologic conditions to 1992 levels and source contributions to 2001 levels, and a second where we set hydrologic conditions to 2001 levels and source contributions to 1992 levels. For model parameters that we assumed were constant over the period of analysis (soil permeability and drainage density), we used the base data available as part of the 2.8 version of the SPARROW model [*Schwarz et al.*, 2006]. For time dependent variables (e.g., observed loading, application rates, and land use conditions), we determined the state conditions for the two study years. Our analysis focused on the incremental total nitrogen yield estimated by the model for over 60,000 river reaches in the contiguous United States. Incremental yield is defined in the SPARROW model as the total flux delivered from the incremental watershed to the reach, normalized by the watershed drainage area with units of kg ha^{-1} yr^{-1}. The model output quantifies nitrogen yield for each nitrogen source considered within the model. A key assumption of our study was to neglect human management practices meant to reduce nitrogen loading (e.g., reductions wastewater point source loadings and the use of Best Management Practices (BMPs) to control nitrogen runoff). This assumption was necessary because of a lack of data for quantifying the impact of these activities, however the results of the analysis should be interpreted in light of this key model limitation.

In the following section we provide details of the study methodology summarized in the previous paragraph. This section is followed by a presentation and discussion of the study results as a means for understanding changes in the spatial distribution of incremental nitrogen yield over the decadal study period. Finally we present a summary and concluding statements from this work, along with suggested extensions to the study materials and methodology that could be accomplished through future research.

2. Materials and Methods

2.1. Model Description

SPARROW is a non-linear regression model that relates measured loadings at monitoring stations to point and nonpoint source loadings, waterbody properties, and watershed attributes in order to predict long-term mean annual instream load for unmonitored reaches [*Schwarz et al.*, 2006]. The measured instream loadings serve as the dependent variable in the regression, while point and nonpoint sources, waterbody properties, and watershed attributes serve as the independent variables. Nitrogen source terms considered by the model include direct sources such as population related sources, atmospheric deposition, fertilizer application and animal waste, as well as indirect sources such as non-agricultural land use. Watershed attributes typically considered in SPARROW include precipitation, temperature, soil permeability and stream density, while waterbody attributes include flow rate, velocity and hydraulic loading (for lakes and reservoirs).

The SPARROW model formulation used in this study states that the mean annual total nitrogen loading observed at reach i is

$$L_i = \Big[\Big\{ \sum_{j \in J(i)} L_j \Big\} A(Z_i^s, Z_i^r; \kappa_s, \kappa_r) + \Big\{ \sum_{n=1}^{N} S_{n,i}\beta_n \Big\} D_n(Z_i^D; \alpha) A'(Z_i^s, Z_i^r; \kappa_s, \kappa_r)\Big]\epsilon_i \quad (1)$$

where the first summation term is the amount of flux that leaves each of the adjacent upstream reaches for reach i where L_j represents the measured or estimated flux leaving the adjacent upstream reach j. The function $A(.)$ represents the loss of mass due to transport to the downstream node of reach i (Figure 1). The vectors Z_i^s and Z_i^r are measured stream and reservoir characteristics, while κ_s and κ_r are corresponding coefficient vectors. If the waterbody i is a stream, then Z_i^s and κ_s define the function $A(.)$ and if the waterbody i is a lake or reservoir, then Z_i^r and κ_r define the function $A(.)$.

The second summation term is the amount of mass originating within the watershed that is delivered to the waterbody i. N is the total number of mass sources, n is an individual mass source, and $S_{n,i}$ is the contribution from mass source n in reach i. β_n is a regression coefficient that represents an array of source-specific coefficients and serves as a conversion factor between the source units and flux units estimated by the model. The function $D(.)$ represents the land-to-water delivery process where α is the estimated vector of coefficients and Z_i^D is the vector of watershed attributes. The term $A'(.)$ represents the instream mass loss for reach i. Finally, ϵ_i is the multiplicative error term defined by the model, which is independent and identically distributed across watersheds in the intervening drainage area between monitoring stations [*Alexander and Smith*, 2005; *Hoos and McMahon*, 2009].

Land-to-water delivery is represented in our model by Equation 2 that represents a first-order decay process

$$D_n(Z_i^D; \alpha) \; = \; \exp(-\alpha' Z_i^D) \tag{2}$$

where α' is an array of the model coefficients that describe the land-to-water delivery for each element in the watershed attribute array, Z_i^D. Instream transport is represented within the model by Equation 3 that models loss as a first-order decay process

$$A(Z_i^s, Z_i^r; \kappa_s, \kappa_r) \; = \; \exp(-{\kappa_s}' T_{i,j}) \tag{3}$$

where ${\kappa_s}'$ is the array of decay coefficients for streams classified by their mean annual flow rates. The decay coefficients are estimations of mass loss per unit stream length. $T_{i,j}$ is an array of waterbody characteristics for the flow path. The first-order decay is modeling losses due to physical processes occurring within the stream such as denitrification and sedimentation [*Alexander and Smith*, 2005]. Finally, for reaches that represent lakes or reservoirs, $A(Z_i^s, Z_i^r; \kappa_s, \kappa_r)$ takes the form of Equation 4 that models nitrogen loss as a settling rate

$$A(Z_i^s, Z_i^r; \kappa_s, \kappa_r) \; = \; \left(1 + \frac{\kappa_r}{q_i^r}\right)^{-1} \tag{4}$$

where κ_r is the reservoir decay coefficient estimated by the model and q_i^r is the areal hydraulic loading (ratio of reservoir outflow to surface area, in units of distance per time).

The model is first calibrated using Equations 1-4 to estimate nitrogen loading for all monitored reaches. The result of the regression is estimates of the coefficient values for β, α, and κ that minimize errors between observed and predicted loadings for monitored reaches. Once these coefficients have been determined, the model is applied to predict nitrogen delivery for unmonitored reaches within the river network. The model results in predictions of incremental and total nitrogen yield for each reach in the river network dataset and for each nitrogen source considered by the model. While there are limitations to the modeling approach used by SPARROW, as we will discuss in greater detail in Section 3.3, we have used the model because it offers a practical blend of process-based and empirical modeling appropriate for regional-scale assessments, where it is difficult to parameterize physical-based models of hydrologic and biogeochemical processes.

In this study we considered four model scenarios: Model I (1992), Model II (2001 Hydrology), Model III (2001 Sources) and Model IV (2001). Model I represents the 1992 conditions. All the sources, land-to-water delivery terms, and mean annual loadings represent the year 1992. Model II represents Model I modified so that the precipitation and mean air temperature are set to 2001 conditions. We use this model scenario to estimate how the hydrologic changes (precipitation and evaporation, which is related to air temperature) impacted nitrogen delivery. Model III represents Model I modified so that the source variables are set to 2001 conditions. This model scenario allows us to estimate loading changes due to changes in nitrogen sources. Finally, Model IV represents the 2001 scenario where all the sources and land-to-water delivery terms represent the year

2001. Our initial approach was to calibrate the model to 1992 and 2001 observed loading data separately, but there were insufficient instream concentration observations available for 2001 to support this approach. Therefore we calibrated the SPARROW model for 1992 conditions and then simulated stream loads for 2001 conditions assuming that the SPARROW model coefficients were unchanged and only state conditions changed. We then used the SPARROW model coefficients from the Model I calibration to also predict loadings for the Model II and Model III scenarios. Our justification for this assumption is that because there were no major departures in state conditions between the decade separating the two study periods, we can assume that the statistical relationship that forms the basis of the model formulation is valid for both years. To verify our assumptions, we evaluated the 2001 model by comparing predicted loadings to the limited set of available observed loadings. A similar approach of calibrating the model for 1992 and then using the calibration coefficients to predict for 2002 was used by *Alexander et al.* [2008] for the Mississippi River Basin.

Once the model has been calibrated and predicted, the incremental yield for each catchment was estimated as

$$\begin{aligned} Yield_i = & \left\{ \sum_{n=1}^{N} S_{n,i}\beta_n \right\} \times \\ & D_n(Z_i^D;\alpha)A'(Z_i^s, Z_i^r;\kappa_s,\kappa_r) \Big/ Area_i \end{aligned} \tag{5}$$

where i is the number of incremental catchments, $Area_i$ is the area of the catchment and $Yield_i$ is the incremental yield in kg ha^{-1} yr^{-1}. Incremental yield gives the yield estimates that originated in that specific catchment without considering the upstream contributions. The incremental yield model output was the primarily focus of this study because we were interested in how loads were delivered to streams under different hydrologic and sourcing conditions.

2.2. Data Preparation

It is typical to use the Enhanced River Reach File 1 (ERF1) dataset as the river network representation for continental-scale SPARROW applications [*Nolan et al.*, 2002]. ERF1 is a digital stream network at a 1:500,000 spatial scale for the contiguous United States that is an improvement over the earlier River Reach File 1 (RF1) dataset [*DeWald et al.*, 1985; *Alexander et al.*, 1999]. The stream network consists of more than 60,000 reaches with mean reach length of 17 km and total reach length of 1 million km. Approximately 2,000 of the river reach features represent large reservoirs with a capacity greater than 6 million m^3 [*Smith et al.*, 1997]. Mean streamflow, stream velocity, and time of travel are included as reach attributes in the dataset and used to model instream loss rates due to sedimentation and denitrification in the SPARROW model [*DeWald et al.*, 1985]. The dataset also includes attributes of stream morphology and hydraulic properties, for example incremental and total drainage area, drainage density, mean water depth, and areal hydraulic load. Mean stream velocity was estimated using a regression based approach that relates stream velocity to long term mean streamflow and stream order. Travel time was then estimated by dividing the reach channel length by the mean stream velocity [*DeWald et al.*, 1985].

Watersheds for each reach within the ERF1 dataset were derived using terrain processing algorithms and the Hydro 1K Digital Elevation Model [*U.S. Geological Survey*, 2000]. Watershed attributes considered in previous applications of SPARROW include precipitation, long term average streamflow, incremental drainage area, soil permeability, slope, drainage density, reach length and mean air temperature [*Smith et al.*, 1997]. However, this previous work found that only soil permeability, drainage density and mean air temperature were statistically significant for explaining the nitrogen fate and transport at the regional spatial scale [*Smith et al.*, 1997]. Later work has shown precipitation to also be a statistically significant watershed attribute in nitrogen modeling using SPARROW [*Hoos and McMahon*, 2009]. Therefore, we selected soil permeability, drainage density, mean air temperature, and precipitation as watershed attributes in our model formulation. Soil permeability was estimated for each watershed using the State Soil Geographic (STATSGO) database [*Schwarz and Alexander*, 1995]. Drainage density is defined as the the ratio of stream length to the drainage area and was calculated from the ERF1 reach length and the area of the Hydro 1K-derived watershed associated with that reach.

We used PRISM (Parameter-elevation Regressions on Independent Slopes Model) data [*PRISM*, 2004] for temperature and precipitation estimates. After creating the maximum and minimum temperature grid from the data available in PRISM, we averaged the maximum and minimum grids to estimate the average temperature for the years 1992 and 2001. After creating the average temperature grid, we estimated temperatures for each ERF1 reach catchment (Figure 4). We created the precipitation grid for the year 1992 as the average from the PRISM data for the years 1991, 1992, and 1993. We followed the same procedure to estimate the average precipitation for the year 2001 by using the years 2000, 2001, and 2002. We took this approach to dampen the variability that can occur in annual data. PRISM precipitation data indicated that 2001 was dryer than 1992 for most of the United States, especially for the West, Midwest and Southwest (Figure 4).

Total nitrogen loading, the dependent variable in the model, was quantified using data from the United States Geological Survey (USGS) National Stream Quality Accounting Network (NASQAN). NASQAN was established by the USGS in 1974 to provide a long-term, systematically collected baseline water chemistry dataset for the nation [*Ficke and Hawkinson*, 1975; *Alexander et al.*, 1996]. We define Total Nitrogen (TN) in this study as nitrate, nitrite, and total Kjeldahl nitrogen in unfiltered samples. Annual loading was estimated by using the Fluxmaster program [*Schwarz et al.*, 2006]. Fluxmaster predicts the continuous daily load from continuous daily observed streamflow and discontinuous nitrogen concentration data by using a nonlinear regression model, and then calculates annual loading by taking averages of the daily estimates of total nitrogen loading. In the estimation process we used stations that have at least 15 observations to reduce estimation uncertainty.

We selected the following Fluxmaster model to relate measured nitrogen concentration to streamflow and other explanatory variables.

$$\begin{aligned} ln(l) &= \lambda_0 + \lambda_1 t + \lambda_2 sin(2\pi t) \\ &\quad + \lambda_3 cos(2\pi t) + \lambda_4 ln(q) + \lambda_5 [ln(q)]^2 + \alpha \end{aligned} \tag{6}$$

where l is the instantaneous nitrogen transport, t is decimal time to account for temporal trends [*Robertson et al.*, 2006], q is instantaneous discharge and λ_0 through λ_5 are regression coefficients. The term α is the sampling and model error assumed to be independent and identically distributed, while the trigonometric terms approximate seasonal variations in transport. The mean annual loading was calculated as

$$L = \frac{1}{365} \sum_{i=1}^{365} exp\Big[\lambda_0 + \lambda_1 t_i + \lambda_2 sin(2\pi t_i)$$

$$+\lambda_3 cos(2\pi t_i) + \lambda_4 ln(q_i) + \lambda_5 [ln(q_i)]^2\Big] V_f \quad (7)$$

where t_i is the i^{th} day of the base year in decimal format, q_i is the average streamflow of the i^{th} day of the year over the multi-year period of the streamflow data, and V_f is the minimum variance bias re-transformation correction factor. The minimum variance unbiased estimator procedure [*Cohn et al.*, 1989] was used in the model. The time period used for estimating the average annual loads was 1970 to 2006, as described in the following paragraph, although for many stations the availability of data was only 1970 to 1995 due to budget reductions and the resulting discontinuation of monitoring for certain stations in 1995.

We obtained total nitrogen concentration, instantaneous streamflow, and daily average streamflow observations for NASQAN sites for the time period 1970-2008. We automated the data retrieval process by using web services provided through the Consortium of Universities for the Advancement of Hydrologic Science, Inc. (CUAHSI) Hydrologic Information System (HIS) [*Maidment*, 2008; *Goodall et al.*, 2008]. We used nitrogen concentration and daily streamflow data from 1970 to 2000 and detrended the water quality and flow model in Fluxmaster to 1992 to estimate long term mean loading. Using this time period, we were able to estimate loading for 354 monitoring stations for our 1992 analysis. We also estimated loading for 2001 with a flow-concentration relationship more closely targeted to the study year using nitrogen concentration and daily streamflow data from 1996 to 2006 and then detrended to 2001. In this way we were able to estimate loading for 122 stations. Again, this 2001 loading dataset was not used for calibration of the SPARROW model because there were too few stations, but instead for comparison of the estimated loading for 2001 from monitoring stations with the predicted loading for 2001 from the SPARROW Model IV simulation.

Nitrogen sources considered in the model are population related sources, atmospheric deposition, fertilizer application, livestock waste, and non-agricultural land. Using publicly available datasets, we quantified each source for the years 1992 and 2001. We used human population estimates for the population related sources, similar to previous SPARROW studies [*Alexander et al.*, 2008]. The assumption is that contribution from population related sources like wastewater effluent, municipal waste, and urban runoff are related to human population. We used county level population estimates from the United States Census Bureau [*U.S. Bureau of Census*, 2010] for 1992 and 2001 and ArcGIS® to calculate the population density for the contiguous United States at 1km resolution. We then estimated the population count for each ERF1 watershed for the years 1992 and 2001.

Atmospheric deposition is a well known and important source of nitrogen, specifically nitrate, to streams [*Jones et al.*, 2001]. We considered wet deposition of inorganic nitrogen (nitrate and ammonia) (kg yr^{-1}) as a measure of atmospheric deposition following the approach used in previous SPARROW model applications [*Smith et al.*, 1997]. We did not include ammonia deposition to avoid double counting of agricultural nitrogen input [*Howarth et al.*, 1996]. Annual estimates from the National Atmospheric Deposition Program [*NADP*, 2010] were used to estimate wet deposition of inorganic nitrogen. These monitoring station estimates were converted to a continuous grid of 1km resolution using an inverse-distance weighting interpolation method. Similar to precipitation data, we averaged 1991, 1992, and 1993 to create the 1992 deposition input dataset. We then summarized the atmospheric deposition loadings for each watershed in the study region.

We followed the same procedure for the 2001 nitrogen deposition, averaging deposition estimates for 2000, 2001, and 2002.

Fertilizer application to both agricultural lands and to urban lands for lawn maintenance is another major source of instream nitrogen. The Association of American Plant Food Control Officials (AAPFCO) [*Ruddy et al.*, 2006] used state total sales rather than county level sales to estimate fertilizer application because county level fertilizer sales data are not reliable. Sometimes multi-county distributors report their sales for a single county rather than the actual counties. Moreover farmers can buy fertilizer from one county and use it in another county, so the spatial distribution of fertilizer sales may be inaccurately represented. Data on the state-level annual sales of commercially produced fertilizer from American Plant Food Control Officials (AAPFCO) and data on the county-level fertilizer expenditure that were obtained from the Census of Agriculture were used to estimate county level nitrogen input from farm fertilizer use in kilograms of nitrogen [*Ruddy et al.*, 2006]. State level non-farm fertilizer sales were estimated from the population data and converted to nitrogen inputs in kilograms of nitrogen [*Ruddy et al.*, 2006]. This county level nitrogen input was then summarized to SPARROW watershed level estimates at 1 km resolution for both 1992 and 2001.

Finally, livestock waste was considered as another source of instream nitrogen loading in our model. Nitrogen input from livestock waste was estimated from county-level livestock population data collected by the Census of Agriculture. *Ruddy et al.* [2006] presented a county level estimate of nitrogen in the livestock waste from confined and unconfined animals. Both recoverable manure from confined animals and unrecoverable manure from confined and unconfined animals were included in this estimation. We estimated SPAR-

ROW watershed level nitrogen input from the county level dataset for 1992 and 2001 at 1km resolution.

We used the National Land Cover Dataset (NLCD) 1992/2001 Retrofit Land Cover Change Product (Fry et all., 2009) to assess land use in the years 1992 and 2001. It is not possible to directly use the NLCD 1992 and 2001 land cover products because the products were generated using different classification methodologies, source image seasonality, georegistration approaches, mapping methodologies, and land use classes [*U.S. Geological Survey*, 2001]. We instead reconstructed two land use datasets from the recently produced NLCD Land Cover Change Product that include major land use types of urban, forest, crop land, grass land and non-agricultural land. Urban land includes areas of low, medium, and high intensity development with a mixture of constructed materials and vegetation [*U.S. Geological Survey*, 2001]. Forest land includes deciduous forest, evergreen forest, and mixed forest; Cropland includes cultivated crops and pasture lands; Pasture lands include both grasses and legumes for livestock grazing; Grassland includes both grassland and shrubs; Non-agricultural land represents the combination of urban, forest, and grasslands. Land use grids for the two study years were used to estimate the total area of each land use type, for each year, and for each of the watersheds.

We used the data described in the previous paragraphs to construct the inputs for the four previously described model scenarios. For Model I, we both calibrated the model and used the calibrated model to predict 1992 loadings. For the remaining three model scenarios, we used the parameters from the Model I calibration to predict loadings. We used the 2.8 version of SPARROW, the latest version of the model at the time of this study.

3. Results and Discussion

3.1. Model Calibration

Model I (1992) resulted in R^2 values for predicted log of flux of 0.885 and R^2 values for predicted log of yield of 0.802 (Table 1, Figure 2). These R^2 values are similar to those obtained in previous SPARROW model applications. For example, *Smith et al.* [1997] reported results from a national SPARROW model for total nitrogen using 414 NASQAN stations in which their R^2 values for log of flux were 0.874 and *Hoos and McMahon* [2009] reported R^2 values for flux and yield were 0.96 and 0.68, respectively for an application of SPARROW for the Southeastern United States. The model residuals (Figure 2) showed some signs of a spatial bias with the highest and lowest loads corresponding to the largest and smallest rivers, suggesting an over-prediction for larger rivers and an under-prediction for small rivers. Some possible bias was also present in specific regions such as the Pacific Northwest, which showed over-predictions in general, and the Midwest, which showed under-predictions in general. SPARROW predicted loading for Model IV (2001) was also compared to actual loading observations estimated using the Fluxmaster program based on the flow and concentration data available from 1996 to 2006 for 122 stations (Figure 3). The R^2 value for this predicted log of flux vs actual log of flux was 0.890 for these stations. It should be noted that the high R^2 value for the 2001 evaluation may be due in part to the 2001 dataset consisting of primarily larger streams when compared to the 1992 dataset, or to a time lag introduced by using long term flow and concentration data for model estimation. Nonetheless, we argue that the evaluation of the model against 2001 observed loads provides a reasonable level of confidence that the model is able to predict loads in this year based on calibrated model coefficients for the 1992 model.

The SPARROW model includes three coefficients, α, β and κ, that are fit during the model calibration process (Table 2). The coefficients have physical meaning in that they allow one to understand losses due to land-to-river and instream transport processes. The α coefficient results for soil permeability, drainage density, mean annual air temperature, and precipitation were consistent with expectations and previous SPARROW results [*Smith et al.*, 1997; *Alexander et al.*, 2008; *Hoos and McMahon*, 2009]. All α coefficients were statistically significant ($p < 0.05$). The magnitude of the α coefficients for the different watershed attributes provides important information on how soil permeability, drainage density, mean annual air temperature and precipitation influence the efficiency with which nitrogen is delivered from the land to waterbodies. As expected, the coefficients for soil permeability and temperature were negative. Highly permeable soils will have higher absorption capacities for nutrients, which act to resist nitrogen transport to streams [*Smith et al.*, 1997]. Temperature has a negative correlation with nitrogen transport because increased temperature results in an increased denitrification rate, therefore decreasing the proportion of nitrogen transported to waterbodies [*Smith et al.*, 1997]. Drainage density, the ratio of stream length to drainage area for a watershed, is positively correlated with nitrogen delivery because having a higher drainage density increases the ability for nitrogen to be delivered from the landscape to waterbodies. Similarly, precipitation is positively correlated with nitrogen transport as higher precipitation indicates higher runoff.

For direct nitrogen sources, the β coefficients account for possible variation in source estimates and will vary between sources [*Smith et al.*, 1997]. For indirect nitrogen sources, the β coefficients represent a source term and provide information about the quantity of nitrogen originating from different sources. The β coefficient for population related sources was 3.57 kg person^{-1} yr^{-1} and was statistically significant ($p < 0.05$). The β coefficients for atmospheric deposition, fertilizer application, and livestock waste production were 0.28, 0.25, and 0.08, respectively, and the β coefficient for non-agricultural land was 310 kg km^{-2} yr^{-1}. The β coefficients for fertilizer application and non-agricultural land were statistically significant ($p < 0.05$), but the β coefficients for atmospheric deposition and livestock waste production were not statistically significant ($p > 0.05$). Land use types urban, crop, forest, and grass land were used as variables in the preliminary model simulations but later dropped based on the following criteria. After considering statistical significance, if a variable was not statistically significant, we used variance inflation factor (VIF) to determine if the multicollinearity was responsible for this. We also used eigenspread values to identify multicollinearity. If the eigenspread value was greater than 100, we dropped that predictor variable from the model [*Schwarz et al.*, 2006]. Our final selection of the predictor variables was based on constant variance and minimum correlation. Cropland was dropped because of the strong collinearity between cropland and fertilizer application. For the same reason only the population related source was included and the urban land source was dropped from the final model.

The model calibration resulted in instream loss coefficients (κ) that were greater for low-flow streams ($Q < 28.3$ m^3 s^{-1}) compared to medium-flow streams (28.3 m^3 s^{-1} $\leq Q \leq$ 283 m^3 s^{-1}). These coefficients were found to be statistically significant for low flow streams ($p < 0.05$), but not for medium streams ($p > 0.05$). This is consistent with previous SPARROW model results and the conclusion that loss rate decreases with increasing stream size [*Alexander et al.*, 2000; *Hoos and McMahon*, 2009]. Model prediction for the reservoir loss coefficient (κ_r) was 7.18 m yr^{-1} in 1992, was statistically significant ($p < 0.05$), and was consistent with *Hoos and McMahon* [2009], which reported a slightly higher reservoir loss rate of 13.1 m yr^{-1} for the Southeastern United States. The standard errors for the coefficients (Table 2) provide a measure of the confidence intervals of the coefficient estimates. The standard error values are in line with prior SPARROW studies and could be used through future work to quantify uncertainty of the model estimates reported in this study.

3.2. Model Predictions

When comparing overall change in loading between Model I (1992) and Model IV (2001), the results indicate a decrease in the median incremental nitrogen yield of 0.67 kg ha^{-1} yr^{-1} (or 8.21%) from 8.16 kg ha^{-1} yr^{-1} in 1992 to 7.49 kg ha^{-1} yr^{-1} in 2001 (Table 3). Table 3 presents various statistical quantities for the predictions including the standard deviation for predicted incremental total nitrogen yield (kg ha^{-1} yr^{-1}) that provide a measure of the distribution of yield results across the entire study area. We have chosen to summarize the incremental nitrogen yield prediction results using the median value because it is less influenced by very high incremental yields predicted for larger river basin systems. That said, we acknowledge that because first-order streams dominate the stream network, the median will be weighted toward changes in smaller streams. *Alexander et al.* [2008] also indicated a slight overall decrease in both simulated loadings based on a SPARROW model, as well as monitoring-based loadings of nitrogen in streams over a similar time period for the Mississippi River Basin. Comparing Model I vs Model II (2001 Hydrology) and then Model I vs Model III (2001 Sources) reveals that this decrease was mainly due to hydrologic differences rather than variability in nitrogen source input. The estimates of median incremental nitrogen yield between Model II and Model I indicate a decrease of 0.71 kg ha^{-1} yr^{-1} (or 8.70%). This decrease is the result of changes in precipitation and mean annual air temperature between 1992 to 2001. The precipitation map [*PRISM*, 2004] indicated a decrease in precipitation in 2001 compared to 1992 (Figure 4), and precipitation is an important nitrogen delivery variable for land-to-water transport. Comparing estimates between Model III and Model I indicates an increase of 0.01 kg ha^{-1} yr^{-1} (or 0.12%) in incremental nitrogen yield (Table 3). This result suggests that only taking into account changes in sources does not account for the changes in incremental nitrogen yield over the period of study. Previous studies in the Mississippi River Basin also showed an increase in nitrogen sources and loading to streams before the early 1980s, but after that time no significant trend was observed [*Goolsby et al.*, 1999; *National Agricultural Statistics Service (NASS)*, 1998; *Alexander and Smith*, 1990; *Council of Environmental Quality (CEQ)*, 1989].

The total nitrogen yield map (Figure 5a) shows the overall incremental yield scenario for 1992 (Model I). This figure shows that the incremental yield estimates are higher in the Upper Mississippi, the Ohio, the southeastern portion of the Missouri, the Tennessee and the Lower Mississippi Basins, but also in the Northeastern and the Pacific West Coast regions. A common characteristic of these regions is that they have either significant agricultural lands or higher human populations, which account for the higher incremental yields. *Alexander et al.* [2008] showed a similar pattern of nitrogen yield for the Mississippi River Basin. The results shown in Figure 5a also suggest that the incremental nitrogen yield estimates are highest in the wettest areas of the country and lowest in the driest areas. Nitrogen can be more easily transported in wetter climates because of higher precipitation, runoff, and discharge. On the other hand, in arid and semiarid climates, low amounts of precipitation and high evaporation rates result in limited runoff and, as a result, reduced nitrogen yields. This may be the reason for the lower nitrogen yield from large populated cities in the Southwestern United States.

Figure 5b shows differences in the incremental yield scenario if hydrologic inputs (mean annual air temperature and precipitation) are set to 2001 conditions but nitrogen source data is held at 1992 conditions (Model I vs Model II). Analyzing the input data shows that the change in precipitation map has a similar pattern as seen in 5b, indicating the importance of precipitation in particular in nitrogen delivery due to nonpoint source pollution transport. Figure 5c shows the results of a scenario where we used 2001 source contribution but 1992 hydrologic conditions to quantify how change in sources only would affect incremental nitrogen yield (Model I vs Model III). Comparing Figure 5c with Figure 5a suggests that the region in the Mississippi Basin with the highest nitrogen yield in 1992 actually saw a decrease in yield when considering only changes in nitrogen sources and controlling for hydrologic/climate changes. Finally Figure 5d presents the overall change in incremental yield during the period 1992 to 2001 (Model I and Model IV). This map shows that some regions of the contiguous United States produced more incremental nitrogen yield, despite the fact that the median yield decreased during the period of analysis. Furthermore, by comparing Figures 5b and 5c to Figure 5d, we can observe changes in incremental nitrogen yield over the study period and determine whether they were due to variability in nitrogen sources or changes in hydrologic/climate conditions. For example, the increase in the Ohio and Tennessee Basins appears to be primarily due to hydrologic changes, while the increase in the northern portion of the Missouri Basin appears to be primarily due to increases in nitrogen sources.

Estimates of median incremental yield for each model scenario, separated by each nitrogen source, are presented in Figure 6. Comparing Model I to Model IV provides an estimate of median incremental yield difference between 1992 and 2001 and indicates a decrease of 0.01 kg ha^{-1} yr^{-1}(2.8%) for population related sources, 0.14 kg ha^{-1} yr^{-1}(15%) for atmospheric deposition, 0.08 kg ha^{-1} yr^{-1}(5.4%) for fertilizer application, 0.03 kg ha^{-1} yr^{-1}(6.1%) for livestock waste production, and 0.22 kg ha^{-1} yr^{-1}(8.0%) for non-agricultural lands. Comparison of Model I vs Model III results, which controls for hydrologic changes, provides insight into how changes in nitrogen sources alone during the study period impacted nitrogen yield to streams. The results show that population related sources had the largest percent increase in nitrogen yield (8.4%), fertilizer application had the second largest percent increase (2.9%), and livestock waste had the smallest (0.7%) increase. Median incremental nitrogen yield from atmospheric deposition and non-agricultural land decreased by 6.6% and 0.15%, respectively. The decrease in nitrogen yield from atmospheric deposition was likely due to the fact that we used wet deposition as an input dataset and, by doing so, this source already accounts for hydrologic changes. Controlling for variability in nitrogen sources (Model I vs Model II) showed population related sources had the largest percent decrease (10%) in nitrogen yield. Thus, while yield from population related sources had an overall decrease of 2.8% when comparing Model I and Model IV, considering also the results from Models II and III suggests that this decrease was primarily due to 2001 being a drier year than 1992. Finally, we noted that the percent yield decrease when controlling for sources (Model I vs Model II) was similar for all sources (ranging between a 7% and 10% decrease for the different nitrogen sources considered). When interpreting these results, it should be noted that even though human population is used as a surrogate for point source and non point urban sources, population is diffuse in nature and the relationship between changes in point sources and changes in population is not ideal but necessary given available information. It should also be noted that fertilizer application and livestock waste were assigned to reaches based on county level data, and this data might exaggerate these sources for many counties outside of the agricultural Midwest. Also, as mentioned previously, management practices intended to reduce nitrogen in waters such as improvements in wastewater treatment, construction of urban retention ponds as Best Management Practices (BMPs), preserving riparian buffer zones, and reductions in automobile emission (if not reflected in NADP measurements) are not considered in this study because we are unaware of accurate national-scale datasets for quantifying the impacts of these activities.

The spatial distribution of changes in yield for each source when comparing Model I vs Model IV suggests that nitrogen yields from population related sources increased in the western United States, Texas, and Florida (Figure 7). However, it also suggests an increase in the Tennessee and Ohio Basins. This is the same region that showed higher precipitation rates between 2001 and 1992, which may explain the higher yield from population sources as precipitation may have caused an increase in nonpoint source pollution from populated areas in this region. Nitrogen yields from atmospheric deposition show a pattern nearly identical to the input dataset of changes in nitrogen deposition over the study period. These data show widespread decreases in wet nitrate deposition in the Midwest and Northeast over the period of analysis [*NADP*, 2010]. Fertilizer yield increased in the West, upper portion of the Missouri Basin, Ohio Basin, and the Northeast, while it decreased in the Southeast and Upper Mississippi Basin. *Alexander et al.* [2008] found that the fertilizer sources decreased in the Upper Mississippi Basin and increased in the Missouri Basin due to expanded corn and soybean production, which agree with our findings. However, *Alexander et al.* [2008] found a decrease in fertilizer application for the Ohio Basin, but we found an increase for a portion of this Basin in our study. The increases due to fertilizer in the Ohio Basin and the upper portion of the Missouri Basin follow the pattern shown in the overall incremental yield change map for 1992 to 2001 (Figure 5d), suggesting that fertilizer increases were the primary source of overall nitrogen yield increase in these regions. Increases due to livestock waste showed a scattered pattern overall with decreases in the lower portion of the Mississippi Basin. Changes due to the non-agricultural land followed the same pattern as precipitation changes because non-agriculture land is well distributed across the study region.

Figure 8 presents source share results for each nitrogen source considered in the model. Source share is defined as the incremental yield for a specific source of nitrogen divided by the total incremental yield for a given watershed. Given the definition of incremental yield (Equation 5), it can be shown that the watershed property term (Z^D) and the in-stream transport terms (Z^s and Z^r) cancel out when calculating source share. In other words, because the transport factors are applied equally to all sources, the changes in climate/hydrology transport factors do not impact the incremental measures of the source shares, which are applied equally for all sources. Therefore, Model I source share results match Model II source share results and, similarly, Model III source share results match Model IV source share results. We presented Model I and Model IV source share results in Figure 8. The model results show an increase in source share contribution from population related sources and livestock waste, and a decrease in source share from atmospheric deposition, fertilizer application, and non-agricultural land. Over the period of analysis population increased by approximately 8% in the contiguous United States [*U.S. Bureau of Census*, 2010] and population related source share increased by 11.5%. The source share for atmospheric deposition decreased by 6.17%, however this decrease may be related to lower precipitation in 2001 compared to 1992, as the model considered wet deposition of inorganic nitrogen (nitrate and ammonia) as an estimate for atmospheric deposition. The decrease in source share from fertilizer application was 3.0%. While the fertilizer application rate data from USDA indicated increases in fertilizer use for some regions of the United States, the model results suggest an overall decrease in source share contribution for fertilizer. This decrease in fertilizer source share could simply be the result of an increase in source share contribution from population related sources. However, this fertilizer source share decrease did not account for changes in farm management practices such as conservation tillage, nutrient management, and BMPs, and nitrogen removal by increasing crop yields, which all likely contributed to reducing nitrogen yield from agricultural lands over the period of analysis. Source share from livestock waste production increased by 1.2%, while source share from non-agricultural land decreased by 1.5%. Both sources showed little change as these sources remain nearly constant over the period of analysis.

To understand the impact of land use type on incremental yield, we estimated incremental yield loading from watersheds that have a dominant land use type (Figure 9 and Table 4). If the land cover of the watershed was more than 90% urban land, we considered it as an urban watershed. We did the same for cropland and grass land. If the land cover was more than 95% forest land, we considered it to be a forested watershed. Results showed that while median incremental yield from urban watersheds experienced only a 0.4% decrease from 1992 to 2001 (Model I vs Model IV results), this was due to an offset between hydrologic changes and source contribution variability. If we consider only changes in hydrology between 1992 and 2001 (Model I vs Model II), the results show a decrease of 7.4% in median incremental yield. If we consider changes in source contribution only over the same time period (Model I vs Model III), the results show an increase of 6.8%. Urban watersheds showed the most significant increase in nitrogen yield due to variability in source contribution after controlling for hydrologic changes, however this result should be interpreted in light of the fact that the study does not account for reductions in point sources of nitrogen that were implemented over the study period. Results also showed an overall decrease in median incremental yield from cropland watersheds of 16% from 1992 to 2001 (Model I vs Model IV). However, this decrease was primarily due to hydrologic changes and not to decrease in source contributions. Incremental yield decrease for cropland due to hydrologic changes (Model I vs Model II) was 12% while decrease due to source contribution variability was 2.1% (Model I vs. Model III). Forested watersheds showed an overall median incremental yield decrease of 5.8% (Model I vs Model IV), but this change was primarily due to hydrologic changes (6.3%; Model I vs Model II) and not decreases in source contribution (0.5%; Model I vs Model III). Finally grassland watersheds showed a 3.8% overall decrease (Model I vs Model IV) with a 7% decrease when considering only hydrologic changes (Model I vs Model II) and 3.5% increase when considering only source contribution variability (Model I vs Model III) in median incremental yield. The median incremental yield from the different land use types were similar to previously published values (Table 4; *Frink* [1991]; *Ritter* [1988]; *Beaulac and Reckhow* [1982]).

3.3. Model Limitations and Assumptions

While we have made note of limitations of our model, data, and methodology throughout this discussion, we highlight the most significant of these limitations and assumptions here. One key limitation is that SPARROW is a semi-empirical model and therefore has inherit differences compared to a more mechanistic modeling approach. However, there are challenges and issues associated with using a more mechanistic model at a national scale including data availability, computational requirements, the need for some empirical assumptions even in mechanistic models, subgrid heterogeneity, and problems of parameter estimation and calibration [*Beven*, 1989; *Smith et al.*, 2008; *Oreskes et al.*, 1994]. Therefore, while there are advantages to using a more mechanistic model for answering our research question, such an approach is not without its own problems and so there is still a role to play for simplified versions of process-based models like SPARROW, a widely applied model both in the United States and abroad [*Elliott et al.*, 2005; *Hoos and McMahon*, 2009]. While we are on one hand making an argument for SPARROW as an appropriate model for our research question, we acknowledge the possibility that SPARROW may be overly simplified for addressing our research question because it does not, for example, include representations for processes such as nitrogen contributions from groundwater, which may be significant for many regions of the country [*Ator and Ferrari*, 1997]. Also previous studies suggest potential problems with the statistical approaches used in SPARROW. For example, *Qian et al.* [2005] suggested a structural weakness in the SPARROW model that may cause spatial autocorrelation. Similarly, the fitted SPARROW model may be biased because it uses statistical inference of a nonlinear regression based on the normality assumption [*Fuller*, 1995].

In this study we assumed that some watershed properties and river conditions were constant from 1992 to 2001 including soil permeability, drainage density, streamflow rates, and stream velocities. Streamflow rates and stream velocities are estimated based on long term average condition. We therefore do not suspect large changes in these attributes over the short time period of analysis, and we believe it to be a justifiable assumption that changes in these parameters from 1992 to 2001 will not significantly alter the findings reported in this study. Another important limitation of the study was our inability to estimate observed nitrogen loading using NASQAN monitoring stations for 2001 at a sufficient number of stations in order to calibrate SPARROW. Due to inadequate observed nitrogen data, we were only able to estimate loading for 1992, when sufficient data were available to estimate flow-concentration relationships. This estimation was based on long term flow condition for 1970-2000 and long term mean load detrended to 1992. To address the limitation of lack of monitoring data for 2001 we considered an alternative approach to only calibrate the SPARROW model for 1992 and then use the calibration coefficients to simulate the loading for 2001. This approach assumes that the model coefficients are unchanged and only state conditions change. *Alexander et al.* [2008] presented a similar approach for Mississippi River Basin SPARROW model study. We kept streamflows and velocities (which are used to estimate travel times) as long term averages in all four model scenarios to be consistent with the long term nitrogen loading estimates. Because we were primarily interested in incremental nitrogen yield, we assumed that precipitation would capture hydrologic changes and that keeping streamflow and velocity as long term averages, to be consistent with instream nitrogen loading data, would be justifiable. An extension to this study would be to identify nitrogen concentration and flow observations from other datasets outside of NASQAN to use in the analysis to obtain better estimates of instream nitrogen loading at monitoring stations for both time periods. In this case, the streamflow and velocity estimates should also be updated to each of the base years of the simulation.

4. Conclusion

The goal of this research was to better understand how the variability in source contributions (anthropogenic and non anthropogenic) and the changes in hydrology/climate affect incremental nitrogen yield within the contiguous United States. We used the SPARROW model, land use change products from the National Land Cover Dataset (NLCD), other source inputs, watershed characteristics and instream nitrogen loading observations to quantify these impacts for more than 60,000 watersheds in the contiguous United States. We built four model scenarios to isolate these changes: Model I was a simulation of 1992 conditions, Model II was a modification of Model I where hydrologic inputs (eg. precipitation and mean air temperature) were set to 2001 conditions, Model III was a modification of Model I where source contribution inputs were set to 2001 conditions, and Model IV was a simulation of 2001 conditions.

The results of this study suggest a decrease of 8.2% in median incremental nitrogen yield from 1992 to 2001 (Model I vs Model IV). The decrease was 15% for atmospheric deposition, 8.0% for non-agricultural land use, 6.1% for livestock waste, 5.4% for fertilizer use, and 2.8% for population related sources. If only changes in nitrogen source contributions were considered (Model I vs Model III), we observe only a small increase (0.1%) in median incremental nitrogen yield. However, if only hydrology related changes were considered (Model I vs Model II), we observe a decrease of 8.7% in median incremental nitrogen yield. Therefore results of this analysis suggest that hydrologic changes – and not decreases in nitrogen source contributions – were primarily responsible for changes in nitrogen yield over the period of analysis. The results confirm previous research findings that suggested significant changes in nitrogen sources to the Mississippi River Basin were not observed after the early 1980s [*Goolsby et al.*, 1999; *National Agricultural Statistics Service (NASS)*, 1998; *Alexander and Smith*, 1990; *Council of Environmental Quality (CEQ)*, 1989]. The results also highlight the importance of precipitation and temperature changes on regional scale nitrogen transport.

The model results suggest decreases in incremental nitrogen yield from some of the highest yield producing areas (e.g., Upper Mississippi Basin). After separating hydrologic and source contributions using the model scenarios we found that, although some of this reduction was due to hydrologic differences between the two years (e.g., 2001 was a drier year than 1992), the change was also due to reductions in source contributions, particularly in the Mississippi Basin. The model results also show some areas that experienced an increase in incremental nitrogen yield over the study period. From the model scenarios we observe in some regions this increase was due to increases in source contribution (e.g., the upper portion of the Missouri Basin), but for other regions this increase was due primarily to differences in hydrology (e.g., the Pacific Northwest).

We found from the model scenarios how each source was dependent on hydrologic vs source contribution changes. For example, although overall median incremental nitrogen yield decreased for population related sources (2.8%; Model I vs Model IV), this overall decrease was due to an offset between an increase in source contribution (8.4%; Model I vs Model III) and a decrease due to hydrologic changes (10%; Model I vs Model II). When incremental nitrogen yield for each source is viewed spatially, we found that changes in fertilizer application in particular was responsible for the overall decrease in nitrogen yield for the Upper Mississippi Basin and the overall increase in nitrogen yield for the upper portion of the Missouri Basin.

We found that source share of the total nitrogen budget for incremental yield increased by 11.5% for population related sources and decreased by 6.17% and 3.0% for atmospheric deposition and fertilizer application, respectively. Source share for livestock waste and non-agricultural land remained nearly constant over the period of analysis. By grouping results for watersheds with dominate land use types, we found that urban watersheds showed the largest percent increase in incremental nitrogen yield (6.8%) and cropland had largest percent decrease (2.1%), after controlling for hydrologic changes. These results suggest that nitrogen from population related sources may becoming a significant contributor of incremental nitrogen yield to streams, however it is important to stress that this study was not able to account for changes in human management practices over the period of analysis that are known to have occur but are not easy to quantify at the scale of this study. Another key limitation of our model was that – because of an insufficient amount of instream nitrogen observation data in and around the year 2001 – we were unable to calibrate the model for 2001 conditions. Therefore we assumed that model coefficients in SPARROW that describe such properties as instream and land-to-water transport were constant between 1992 and 2001. However, we did evaluate the 2001 model predictions against the available instream nitrogen loading data and model showed a good fit to these observed data. One possible extension of this work would be to identify other reliable water quality datasets that could be used to improve estimates of instream nitrogen loading in 2001 so that the SPARROW model can be re-calibrated for the 2001 model scenario.

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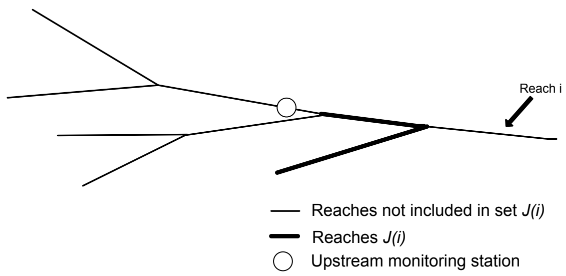

Figure 1. Schematic illustration for SPARROW reaches where $J(i)$ is the set of adjacent reaches upstream of reach i.

R^2 flux	0.885
R^2 yield	0.802
Mean square error	0.404
Number of observations	354

Table 1. Results of Model I (1992) calibration

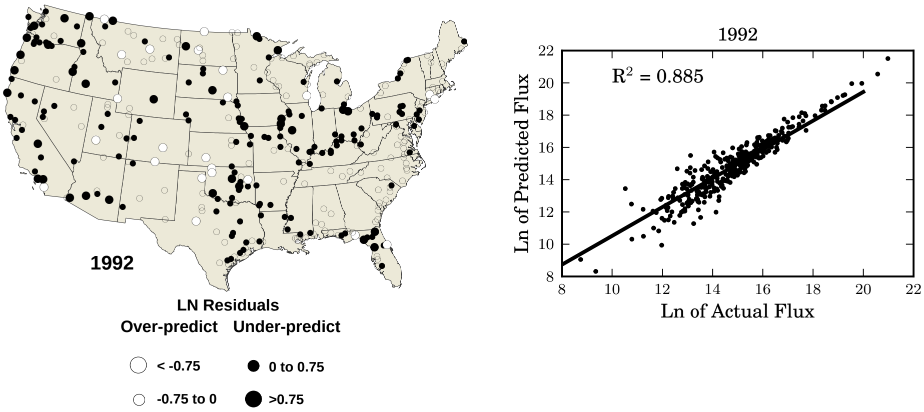

Figure 2. Predicted flux from Model I (1992) versus actual flux of total nitrogen for the 354 monitoring stations for 1992

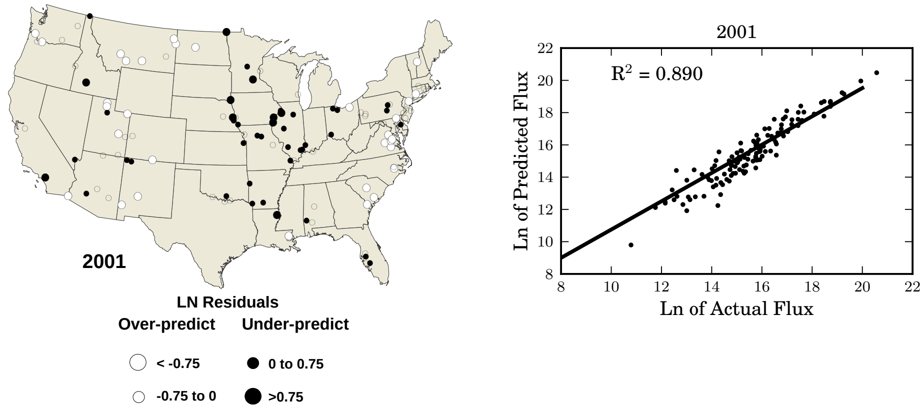

Figure 3. Predicted flux in Model IV (2001) versus actual flux of total nitrogen for the 122 monitoring stations for 2001

Parameter	Units	Coefficient	Standard Error	P-Value
Nitrogen Sources, β				
Population related Sources	kg person^{-1} yr^{-1}	3.565	0.801	<0.05
Atmospheric deposition	dimensionless	0.282	0.179	0.117
Fertilizer application	dimensionless	0.248	0.046	<0.05
Livestock waste production	dimensionless	0.083	0.074	0.260
Non Agricultural Land	kg km^{-2} yr^{-1}	310.05	53.68	<0.05
Land to water delivery, α'				
Soil permeability	in hr^{-1}	-0.097	0.017	<0.05
Drainage density	km^{-1}	2.006	0.479	<0.05
Mean annual air temperature	°C	-0.061	0.009	<0.05
Precipitation	cm	0.009	0.001	<0.05
In-stream decay, κ'				
κ_1 $(Q \leq 28.3\, m^3\, s^{-1})$	day^{-1}	0.226	0.031	<0.05
κ_2 $(28.3\, m^3\, s^{-1} < Q < 283\, m^3\, s^{-1})$	day^{-1}	0.059	0.025	<0.05
Reservoir decay, κ_r	m yr^{-1}	7.182	1.938	<0.05

Table 2. Resulting model coefficients for the SPARROW model from the Model I (1992) calibration.

Incremental Nitrogen Yield (kg ha^{-1} yr^{-1})	**10th**	**25th**	**50th**	**75th**	**90th**	**Mean**	**Standard Deviation**
Model I (1992)	2.9	4.52	8.16	14.11	24.64	17.91	603.51
Model II	2.66	4.19	7.45	13.12	22.98	16.66	528.83
Model III (2001 Sources)	2.95	4.64	8.17	14.18	24.63	18.03	600.64
Model IV (2001)	2.71	4.3	7.49	13.2	22.91	16.8	525.7
Total Change (kg ha^{-1} yr^{-1}) Model IV - Model I	-0.19	-0.22	-0.67	-0.91	-1.73	-1.11	-
Total Change (%) Model IV - Model I	-6.55	-4.87	-8.21	-6.45	-7.02	-6.20	-
Change due to Hydrology (%) Model II - Model I	-8.28	-7.30	-8.70	-7.02	-6.74	-6.98	-
Change due to Sources (%) Model III - Model I	1.72	2.65	0.12	0.50	-0.04	0.67	-

Table 3. Distribution of incremental total nitrogen yield (kg ha^{-1} yr^{-1}) for the different model scenarios.

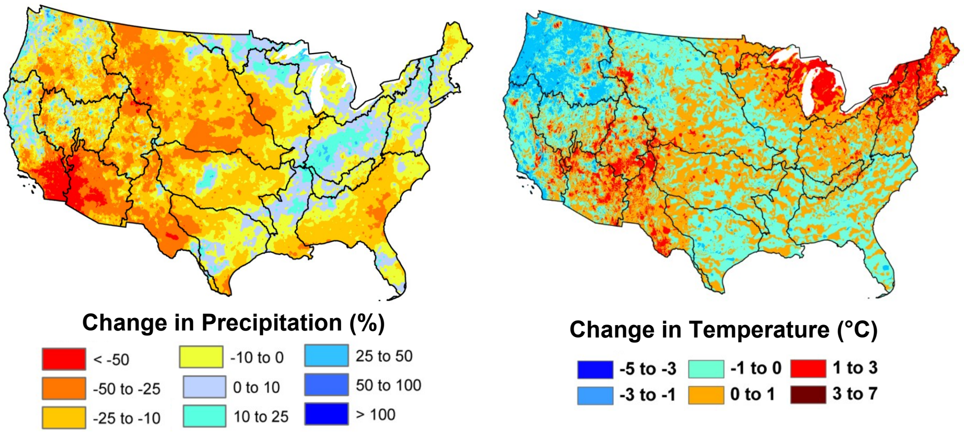

Figure 4. Percent change in precipitation and temperature between the input datasets used for the 1992 and 2001 models.

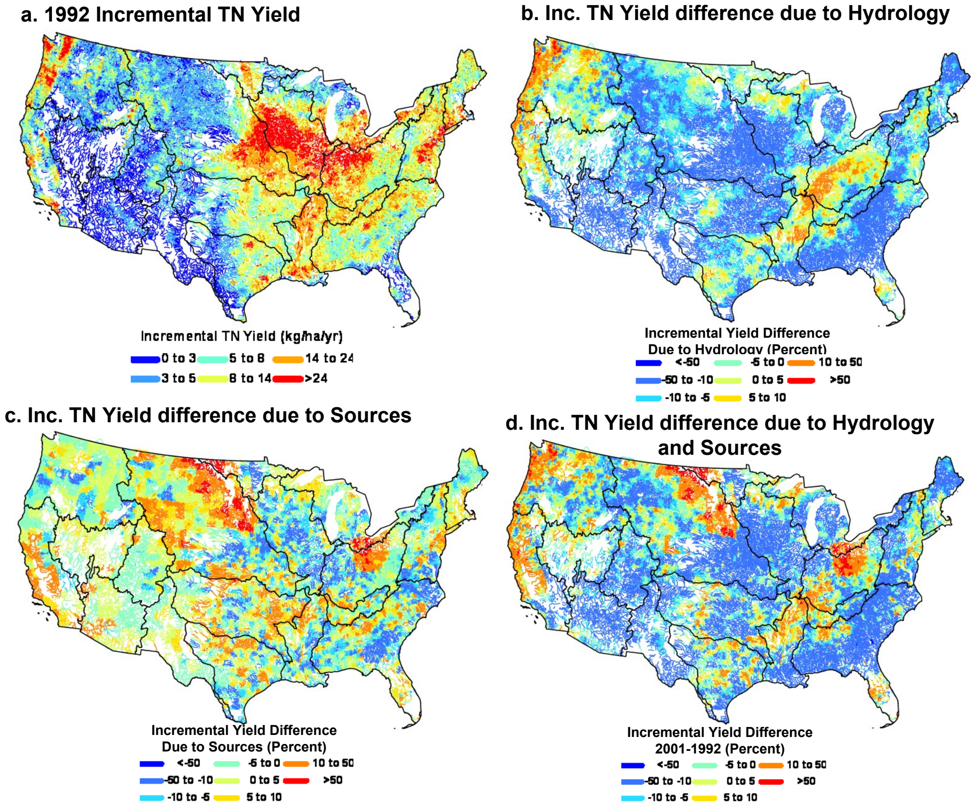

Figure 5. The top left map (a.) shows incremental nitrogen yield scenario in 1992. The top right map (b.) shows the percent difference in incremental nitrogen yield because of change in hydrology between 1992 and 2001. The bottom left map (c.) shows the percent difference in incremental nitrogen yield due to the change in source contribution between 1992 and 2001. The bottom right (d.) map shows the percent difference of incremental nitrogen yield due to overall change between 1992 and 2001.

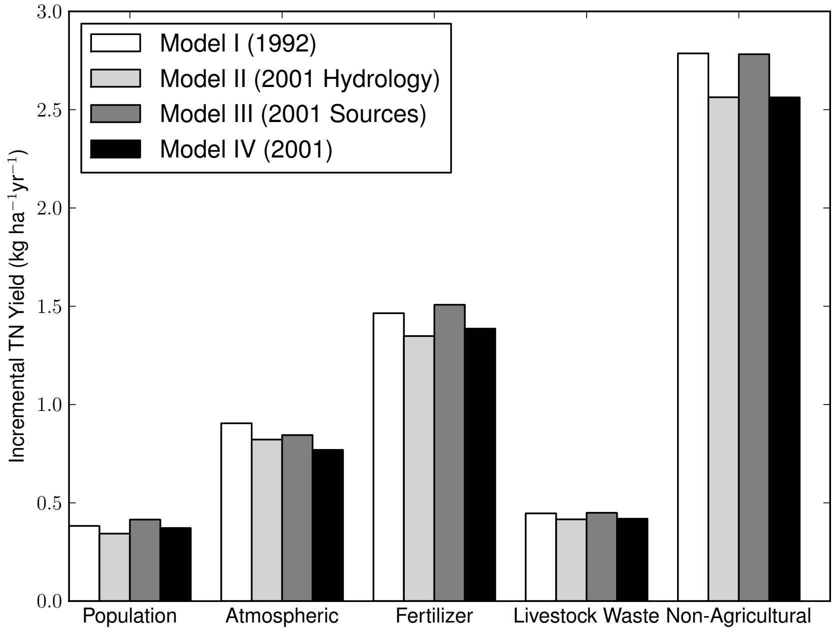

Figure 6. Median incremental total nitrogen yield (kg ha^{-1} yr^{-1}) for different sources of nitrogen and for each model scenario.

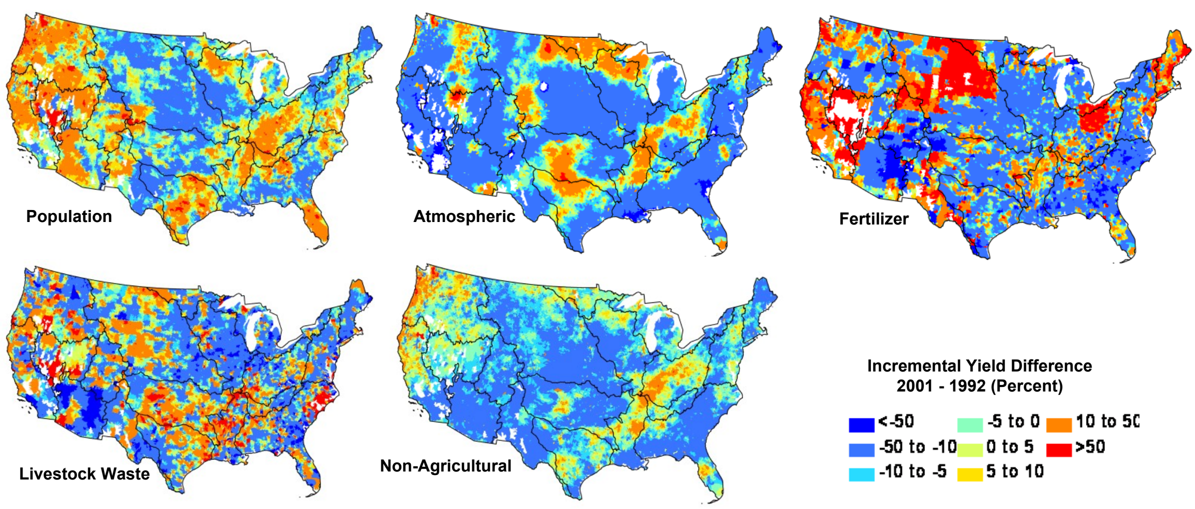

Figure 7. Percent difference of incremental nitrogen yield between Model I (1992) and Model IV (2001) for different sources of nitrogen

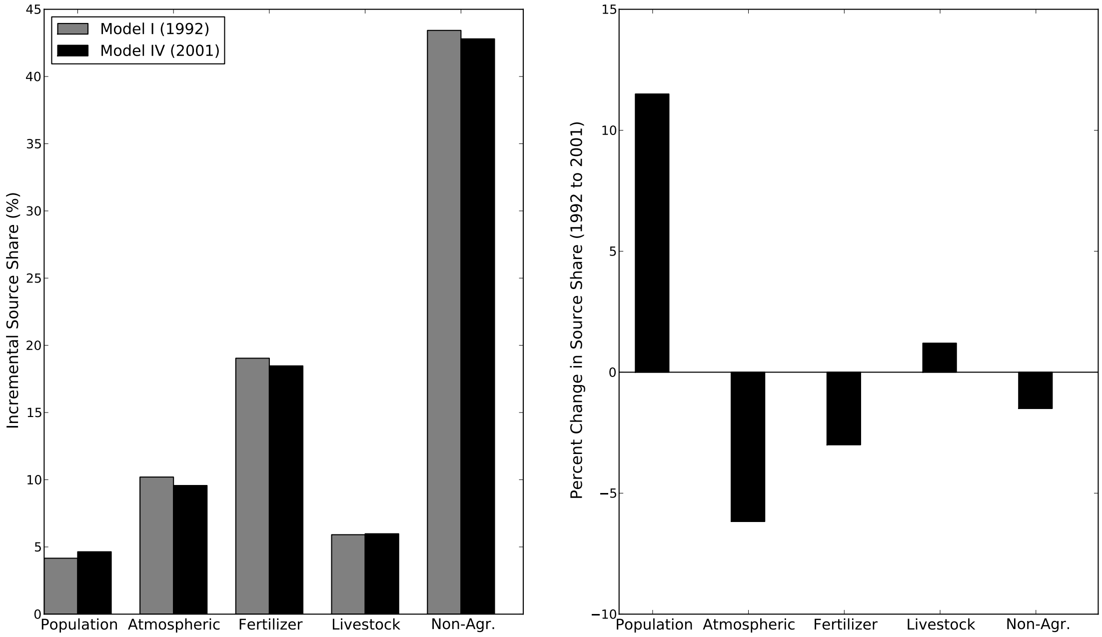

Figure 8. Comparison of median incremental total nitrogen source share (%) contribution for Model I (1992) and Model IV (2001).

Model No	Watershed Type[1]	Distribution of Incremental Yield (kg ha^{-1} yr^{-1})					Range of Yield values from Literature[2] (kg ha^{-1} yr^{-1})
		10^{th}	25^{th}	50^{th}	75^{th}	90^{th}	
Model I (1992)	Urban	12.61	17.82	32.2	66.65	113.75	1.6 - 38.5
	Forest	5.49	7.77	11.13	16.38	30.21	0.1 - 10.8
	Crop	8.09	13.61	23.73	32	40.83	0.8 - 79.6
	Grass	1.73	2.49	3.67	5.26	8.26	0.1 - 30.8
Model II (2001 Hydrology)	Urban	12.16	15.17	29.81	60.31	119.84	1.6 - 38.5
	Forest	5.26	7.42	10.43	16.09	30.72	0.1 - 10.8
	Crop	7.67	12.75	20.81	27.29	34.91	0.8 - 79.6
	Grass	1.49	2.27	3.41	4.9	7.52	0.1 - 30.8
Model III (2001 Sources)	Urban	13.46	19.24	34.38	55.35	105.06	1.6 - 38.5
	Forest	5.62	7.82	11.08	16.37	29.92	0.1 - 10.8
	Crop	7.91	14.21	23.23	30.93	41.63	0.8 - 79.6
	Grass	1.73	2.55	3.8	5.45	8.61	0.1 - 30.8
Model IV (2001)	Urban	13	17.09	32.07	49.69	111.31	1.6 - 38.5
	Forest	5	7.5	10.49	16	29.96	0.1 - 10.8
	Crop	7.72	13	19.89	26.21	35.31	0.8 - 79.6
	Grass	2	2.32	3.53	5.1	7.86	0.1 - 30.8

Table 4. Incremental total nitrogen yield (kg ha^{-1} yr^{-1}) for different model scenarios and literature estimates for watersheds with dominate landuse type in the United States.
[1]The land cover types are based on the following percentage of land use area in SPARROW watersheds: urban(>90%), forest (>95%), crop land (>90%), grass (>90%)
[2] Literature reported values for incremental total nitrogen yield [*Frink*, 1991; *Ritter*, 1988; *Beaulac and Reckhow*, 1982].

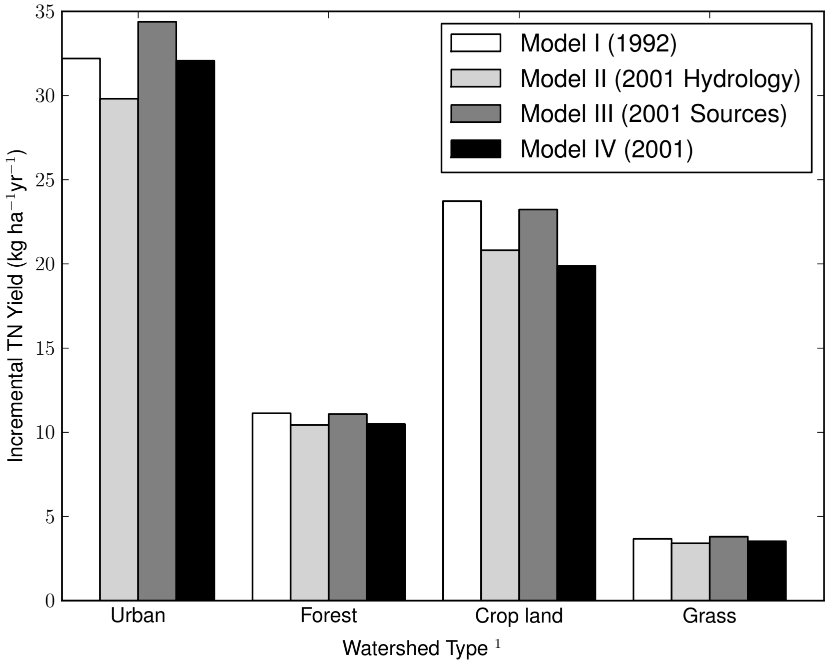

Figure 9. Median incremental total nitrogen yield (kg ha^{-1} yr^{-1}) for major land use types in the United States for different model scenario.

[1]The land cover types are based on the following percentage of land use area in SPARROW watersheds: urban(>90%), forest (>95%), crop land (>90%), grass (>90%)