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Estimating effects of reforestation on nitrogen and phosphorus load reductions in the Lower Yazoo River Watershed, Mississippi

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ABSTRACT

Surface water quality in the Lower Mississippi River Basin (LMRB) and the adjacent Gulf of Mexico has degraded over the past several decades primarily due to deforestation to agricultural lands and the loss of wetlands. This study investigated the benefits of reforestation upon nitrate–nitrogen ($NO_3^- - - -N$) and orthophosphate (PO_4^{3-}) load reductions in the Lower Yazoo River Watershed (LYRW) within the LMRB using the BASINS-HSPF model. The model was calibrated and validated with available experimental data prior to its applications. Two simulation scenarios were then performed: one was chosen to predict the $NO_3^- - - -N$ and PO_4^{3-} loads without reforestation and the other was selected to estimate the impacts of reforestation upon $NO_3^- - - -N$ and PO_4^{3-} load reductions following the conversion of 25, 50, 75, and 100% of the agricultural lands (with most lands near or in the batture of the streams) into forests. In general, an increase in forests reduced $NO_3^- - - -N$ and PO_4^{3-} loads and occurred because forest soils enriched in organic matter absorb water and nutrients and reduce the surface water runoff. Overall, a two-fold increase in forest land would result in approximately two-fold decrease in annual average $NO_3^- - - -N$ and PO_4^{3-} loads. On average, over a 10-year simulation, the specific $NO_3^- - - -N$ and PO_4^{3-} load reductions were, respectively, 0.06 and 0.004 ton/ha/y. Although the annual average $NO_3^- - - -N$ and PO_4^{3-} loads always decreased with increasing forest land conversion, the optimal specific $NO_3^- - - -N$ and PO_4^{3-} load reductions were found at a 75% reduction of agricultural land for the simulation conditions used in this study. Additionally, the annual average $NO_3^- - - -N$ load was about 16 times higher than that of PO_4^{3-} in the LYRW. This study suggests that reforestation in or around the batture of streams is a beneficial practice for $NO_3^- - - -N$ and PO_4^{3-} load reductions.

1. Introduction

Water quality in the Mississippi River Basin (MRB) and the adjacent Gulf of Mexico (GOM) has degraded over the past several decades primarily due to deforestation and wetland loss. Eutrophication of GOM with nitrogen (N) discharged from the MRB has been well documented (Goolsby, 2000; Rabalais et al., 2002; Mitsch et al., 2006; Alexander et al., 2008). The use of N fertilizer for agricultural practices has been dramatically increased in the MRB since 1950s and a significant amount of the excessive nitrate–N ($NO_3^- - - -N$) are routed through drainage tiles, ditches, streams, and rivers into the GOM (Goolsby, 2000; National Research Council, 2000; Mitsch et al., 2001; Bianchi et al., 2010). Goolsby (2000) estimated the N source and flux to the GOM from the MRB and found that the major cause for the eutrophication of GOM is the increase in N delivery, especially $NO_3^- - - -N$. The concentrations of $NO_3^- - - -N$ have increased several folds during the past 100 years in streams from the MRB, and the annual delivery of $NO_3^- - - -N$ from the MR to the GOM has nearly tripled since the late 1950s (Goolsby et al., 1999). Goolsby (2000) also found that the average concentration of $NO_3^- - - -N$ in the MRB is 1.45 g N/L and its export to the GOM is about 1,000,000 ton/y.

It has been estimated that the major excess nutrients are derived from corn and soybean farms runoff in the Midwest of the USA, animal feedlots, sewage treatment plants, and industrial sources (MART, 2006; Mitsch et al., 2001; Bianchi et al., 2010). The mean annual flux is about 1.2 million metric tons of N and 0.15 million metric tons of phosphorus (P) (Aulenbach et al., 2007). Of which about 22% for N and 34% for P are from point sources (MART, 2006). The increase in delivery of $NO_3^- - - -N$ can enhance the production of organic carbon in the GOM. Rabalais et al. (1999) assessed that an atom of N from the Mississippi River is recycled

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about four times in the GOM before it is lost from the water column. Therefore, an approximately 0.95 million metric tons of $NO_3^- - - -N$ discharged annually from the MRB (Goolsby et al., 1999) could potentially produce more than 20 million metric tons of organic carbon annually in the GOM, which could lead to the dissolved oxygen (DO) depletion in water column and to the elevated extent and severity of seasonal hypoxic zones (DO < 2 mg/L). The hypoxic zone will cause stress or death in bottom-dwelling organisms that cannot leave the zone (Rabalais et al., 2002; Lohrenz et al., 2008).

While the flux of N may have decreased since 1990 in the MRB, the flux of P has remained steady (Turner et al., 2008). Although $NO_3^- - - -N$ is traditionally considered as the most important nutrient for phytoplankton growth, phosphorus could be a limiting factor for phytoplankton (Sylvan et al., 2006). When oxygen concentration in bottom water decreases, phosphate can be released rapidly from the sediment (Sutula et al., 2004; Howarth and Marino, 2006). If the phosphate can be mixed upward to the pycnocline (the layer where density change with depth is at a maximum), it can fuel phytoplankton production in areas that would otherwise be considered P-limited. Therefore, the release of P from sediments may have a significant role in maintaining primary production (Sutula et al., 2004).

Deforestation to agricultural lands in the MRB has tremendous impacts on nutrient loads to aquifers, streams, and coastal waters. A 500% increase in agricultural N fertilizer application from 1950–1970 to 1980–1996 is the primary contributor to a 200% increase in $NO_3^- - - -N$ export from the MRB to the GOM (Goolsby and Battaglin, 2001), while states within the U.S. Corn Belt are major contributors to nutrient loading (Schilling and Zhang, 2004). Schilling and Zhang (2004) reported that $NO_3^- - - -N$ export from the Raccoon River catchment, Iowa is the highest out of 42 Mississippi River sub-catchments evaluated in a GOM hypoxia study. Alexander et al. (2008) reported that about 70% of the N and P exported to the GOM is from agricultural practices such as the elevated fertilizer application, the extended corn and soybean growth, and the increased surface water runoff (Howarth et al., 2002; Zhang and Schilling, 2006). For the N load into the GOM, corn and soybean cultivation accounts for 52% and atmospheric deposition accounts for 16%. For the P load into the GOM, pasture and rangelands account for 37%, corn and soybean cultivation 25%, other crops 18%, and urban sources 12%. These authors further stated that the amount of in-stream P and N loads into the GOM increases with stream size although reservoir trapping of P causes large local- and regional-scale differences.

Despite numerous efforts devoted to investigating the relationships between the ecological and environmental consequences of deforestation and the benefits of reforestation in the MRB (Harris, 2006), our literature search revealed that the impacts of reforestation on nutrient loads in the MRB are still poorly understood. With an increased appreciation of the importance of drinking water quality to public health and raw water quality to terrestrial life, there is a great need to further examine these issues. Since the dynamics of nutrient variation and load reduction in a given watershed are complex, it is very difficult to quantify them by experimentation alone for different types of land uses, for a variety of soil and hydrological conditions, and for all possible combinations of surficial driving forces. Therefore, a need exists to employ the modeling approach for this purpose.

Recently, we have published a study regarding the impacts of reforestation upon sediment load and water outflow in the Lower Yazoo River Watershed (LYRW), Mississippi (Ouyang et al., 2013). In this companion study, our focus was to estimate potential impacts of reforestation upon $NO_3^- - - -N$ and PO_4^{3-} load reductions in the same watershed. Specific objectives of this companion study were to: (1) extend the site-specific BASINS-HSPF model for LYRW to include the $NO_3^- - - -N$ and PO_4^{3-} load predictions; (2) calibrate and validate the $NO_3^- - - -N$ and PO_4^{3-} components of the model using the field measured data; and (3) apply the model to investigate the role of reforestation (i.e., a conversion of agricultural land near bank into forests) on $NO_3^- - - -N$ and PO_4^{3-} load reductions in the LYRW.

2. Materials and methods

2.1. Study sites

The LYRW is located in the south Yazoo River Basin (YRB), Mississippi, USA (Fig. 1). This watershed consists of 61% forest land and 31% agriculture land with soil types of sand, loam, and clay. Surface water pollution within the YRB includes excess nutrients, sediments, heavy metals, and herbicides, which are the results of storm water runoff, discharge from ditches and creeks, groundwater seepage, aquatic weed control, naturally-occurring organic inputs, and atmospheric deposition (Nett et al., 2004; Pennington, 2004; Aulenbach et al., 2007; Alexander et al., 2008; Shields et al., 2008). An elaborate description of the study site can be found in our previous study (Ouyang et al., 2013).

2.2. Model development and data acquisition

The site-specific BASINS-HSPF model developed for hydrological processes and sediment load for the LYRW from our previous study (Ouyang et al., 2013) was extended to incorporate the $NO_3^- - - -N$ and PO_4^{3-} transport and load in this companion study. The HSPF modules such as PERLND, IMPLND, and RCHRES used in this study were the same as those used in the previous study. Additionally, those sub-modules related to $NO_3^- - - -N$ and PO_4^{3-} transport and load, including PQUAL from PERLND, IQUAL from IMPLND, and RQUAL from RCHRES, were also turned on. The PQUAL sub-module is used to simulate the fate, transport and load of nutrients (e.g., $NO_3^- - - -N$ and PO_4^{3-}) and other pollutants frompervious lands into streams under different land use patterns, whereas the IQUAL sub-model is used to simulate the fate, transport and load of nutrients and other pollutants from impervious lands into streams. The RQUAL sub-module is used to simulate the routing of nutrients and other pollutants within reaches. The physicochemical processes of N and P used in HSPF model include nitrification, denitrification, sorption, precipitation, leaching, and loading. Detailed information about the structure, functions, and limitations of these modules and sub-modules is beyond the scope of this study, but can be found from HSPF User's Manual and elsewhere (Donogian et al., 1984; Bicknell et al., 2001; Ouyang et al., 2013; Sommerlot et al., 2013). Fig. 1 shows the modeled domain for the LYRW used in this study.

Data collection for the LYRW, including watershed descriptions, meteorological conditions, hydrologic processes, and sediment load, were presented in our previous study (Ouyang et al., 2013). The new dataset required for this companion study is the $NO_3^- - - -N$ and PO_4^{3-} concentrations from the LYRW outlet. The data were downloaded from the USGS station (07288955) in Yazoo River BL Steele Bayou near Long Lake, Mississippi (Fig. 1).

2.3. Model calibration and validation

The hydrologic and sediment components of the BASIBS-HSPF model for the LYRW have been calibrated and validated in our previous study (Ouyang et al., 2013). In this study, we only need to

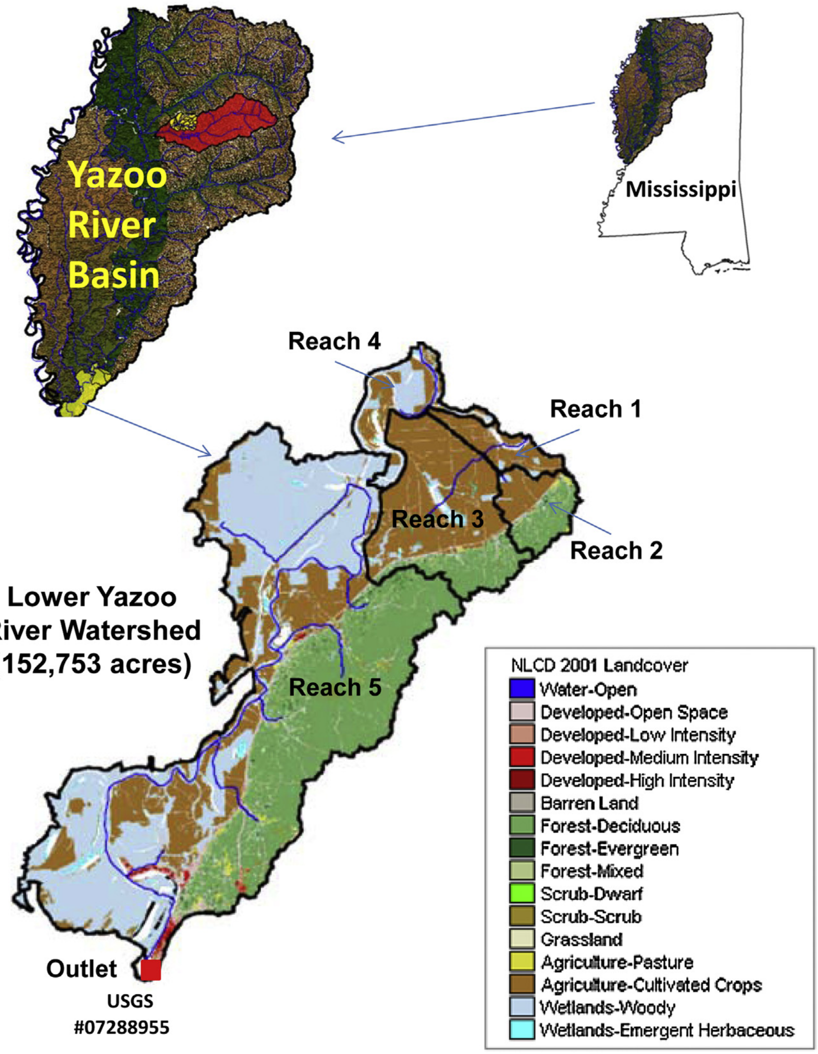

Fig. 1. Location and land use cover for the Lower Yazoo River Watershed.

calibrate and validate the NO_3^- – – –N and PO_4^{3-} components of the model. The calibration period extended from January 1, 2000 to December 31, 2005, whereas the validation period spanned from January 1, 2006 to December 16, 2009. To assure fewer uncertainties in the calibration process, we adjusted the following seven water quality parameters: MON-ACCUM, MON-IFLW-CONC,

Table 1
Calibrated input parameter values for nitrate–N and phosphate simulations.

Land use	MON_ACCUM Nitrate–N	MON-SQOLIM	MON-IFLW-CONC	MON-GRND-CONC	POTFW Orthophosphate
Urban or built-up	0.65–0.75	0.36–0.72	0.65–0.75	0.35–0.40	0.25
Agricultural land	0.60–2.40	1.26–3.16	0.60–2.40	0.30–0.90	0.25
Forest land	0.54–0.58	0.09–0.13	0.54–0.58	0.25	0.25
Wetlands/water	0.54–0.58	0.09–0.13	0.54–0.58	0.25	0.25
Barren land	0.60–2.40	1.26–3.16	0.60–2.40	0.30–0.90	0.25

	MALGR	PHYSET
Nitrate–N	0.001	0.02
Orthophosphate	0.001	0.02

Parameter	Definition
MON_ACCUM	Monthly values of accumulation rates of nutrients at start of each month
MON-SQOLIM	Monthly values limiting storage of nitrate–N and phosphate at start of each month
MON-IFLW-CONC	Monthly values of conc of nitrate–N and phosphate in interflow outflow at start of month
MON-GRND-CONC	Monthly values of conc of nitrate–N and phosphate in groundwater at start of each month
POTFW	Washoff potency factor for orthophosphate
MALGR	Maximal unit algal growth rate for phytoplankton
PHYSET	Rate of phytoplankton settling

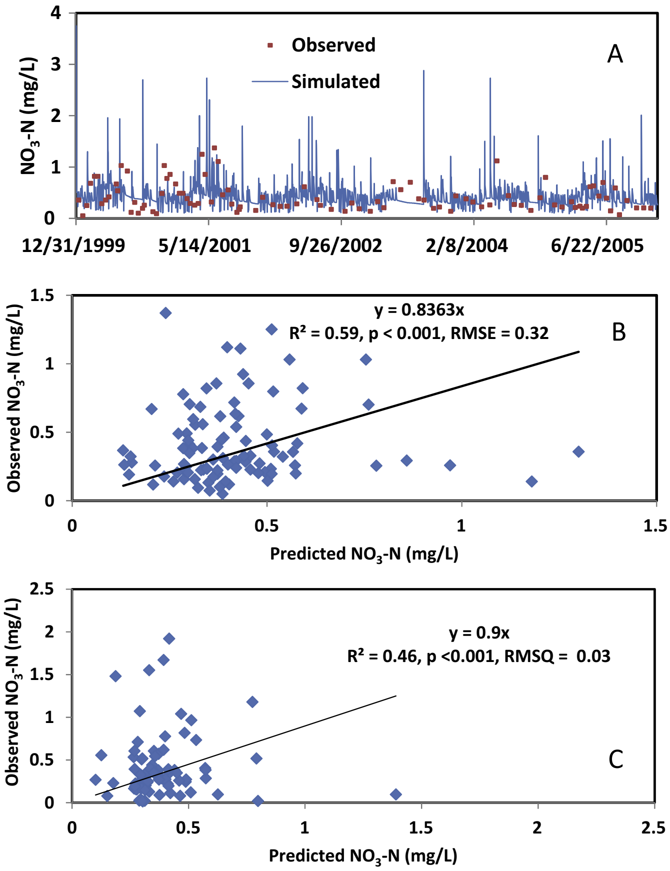

Fig. 2. Daily graphic comparison (A), calibration (B), and validation (C) for nitrate–N in the Lower Yazoo River Watershed.

MON-GRND-CONC, MON-SQOLIM, POTFW, MALGR, and PHYSET, which were defined in Table 1. These parameters are sensitive to NO_3^-– – –N and PO_4^{3-} predictions (Donogian et al., 1984).

Figs. 2 and 3 showed the NO_3^-– – –N and PO_4^{3-} calibrations for the LYRW, which were accomplished by adjusting the aforementioned parameters (Table 1) to match the predicted NO_3^-– – –N and PO_4^{3-} concentrations with the field measurements. The calibration period was from January 1, 2000 to December 31, 2005. The daily peak concentrations from model predictions matched reasonably well graphically with those from field observations for NO_3^-– – –N (Fig. 2A) and for PO_4^{3-} (Fig. 3A). With the values of R^2 and p, respectively, equal to 0.5889 and 3.09E-20 for NO_3^-– – –N (Fig. 2B) and 0.6407 and 4.61E-23 for PO_4^{3-} (Fig. 3B), we concluded that the reasonably good agreements were obtained between the model predictions and the field observations.

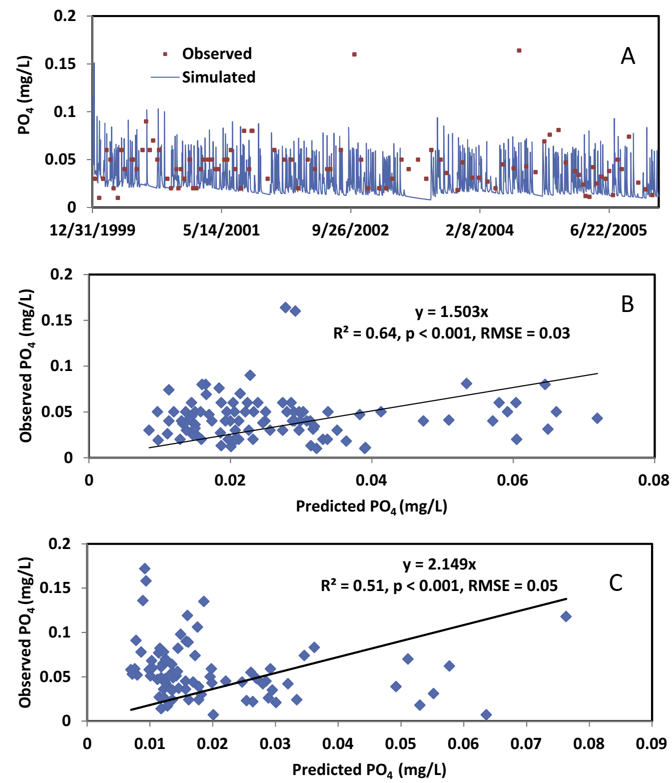

Fig. 3. Daily graphic comparison (A), calibration (B), and validation (C) for orthophosphate in the Lower Yazoo River Watershed.

Validations of the calibrated NO_3^-– – –N and PO_4^{3-} components of the model were given in Figs. 2 and 3C. These figures compared NO_3^-– – –N and PO_4^{3-} concentrations between field observations and model predictions over a time period from January 1, 2006 to December 31, 2010. With the values of p equal to 1.93E-12 for NO_3^-– – –N and 5.23E-14 for PO_4^{3-}, we concluded that the reasonable good agreements were obtained between the model predictions and the field observations. It should be noted that the observed data were not the daily or monthly measured data and they are intermittent, while the predicted data were on a daily base. The model could be vigorously calibrated and validated if the daily NO_3^-– – –N and PO_4^{3-} observed data were available.

2.4. Simulation scenarios

Two simulation scenarios were performed in this study. The first scenario (base scenario) was chosen to predict the NO_3^-– – –N and PO_4^{3-} loads without reforestation (i.e., 0% land use conversion). In this scenario, all of the simulation conditions

Table 2
Forest land increment, specific water outflow attenuation, and specific nutrient load reductions among five different percentages of land use conversion from agricultural land into forest land.

Percentage conversion of agricultural land to forest land (%)	Forest land (ha)	Increase in forest land (ha)	Average annual water outflow (m^3)	Specific water outflow attenuation (m^3/ha/y)	Average annual nitrate–N load (ton)	Specific nitrate–N load reduction (ton/ha/y)	Average annual phosphate load (ton)	Specific phosphate load reduction (ton/ha/y)
0	33,047	0	390,901,025	0	3991	0	252	0
25	35,172	2125	390,452,487	211.075	3870	0.0568	243	0.00428
50	37,297	4250	389,891,815	237.460	3749	0.0570	234	0.00429
75	39,422	6375	389,106,873	281.434	3553	0.0686	219	0.00526
100	41,547	8500	388,658,335	263.844	3492	0.0587	216	0.00426

and input parameter values were the same as those used during model validation. The second scenario was selected to estimate the effects of reforestation upon NO_3^- – – –N and PO_4^{3-} load reductions by converting 25, 50, 75, and 100% of the agricultural lands into the forests. These land use conversions occurred in or near the batture of the streams (Fig. 2). Table 2 shows the area of forest land increase in the LYRW for each percent conversion. In this second scenario, all of the simulation conditions and input parameter values were the same as those used in the first scenario except for the land use conversions. Therefore, comparison of the simulation results from the two scenarios (i.e., with and without reforestations) allowed us to evaluate the effects of reforestation upon the daily, seasonal, and annual NO_3^- – – –N and PO_4^{3-} transport as well as their load reductions. The simulation period was 10 years starting at the first day of 2000 and terminating at the end of 2009 for each scenario, and the agricultural land conversion only occurred in Reach 5 (Fig. 1). It should be noted that the HSPF model includes the following five land uses: urban or built-up, agricultural, forest, wetland/water, and barren land. Each land use has a forest fraction value as input in HSPF. For example, the forest fraction is 0.1 for urban or built-up land, 0 for barren land, 0.1 for agricultural land, and 0.7 for forest land. A conversion of one land use to another will change the forest fraction in the HSPF model (Ouyang et al., 2013). All of the input parameter values for hydrology and sediment components are provided in our previous report (Ouyang et al., 2013). Table 1 lists the major calibrated input parameter values used for NO_3^- – – –N and PO_4^{3-} simulations in this study.

3. Results and discussion

3.1. Nutrient load without reforestation

Changes in daily rainfall event, water discharge, and NO_3^- – – –N and PO_4^{3-} concentrations, which occurred from 2000 to 2009 without reforestation (base scenario), are shown in Fig. 4. The rainfall event was obtained from nearby weather station and further computed to represent the average watershed conditions, whereas the water discharge was attained from our

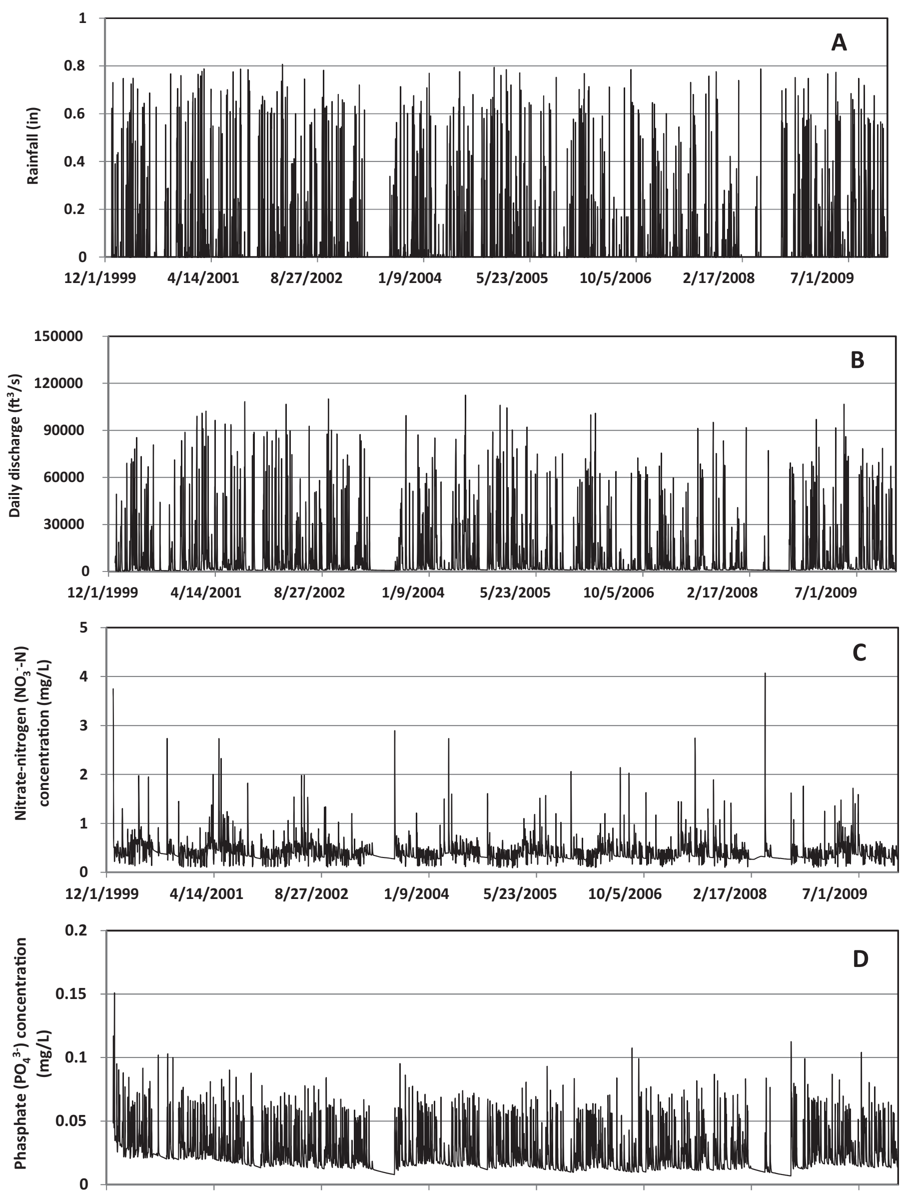

Fig. 4. Daily rainfall (A) as well as predicted water flow (B), nitrate–N (C), orthophosphate (D) concentrations from the base simulation scenario.

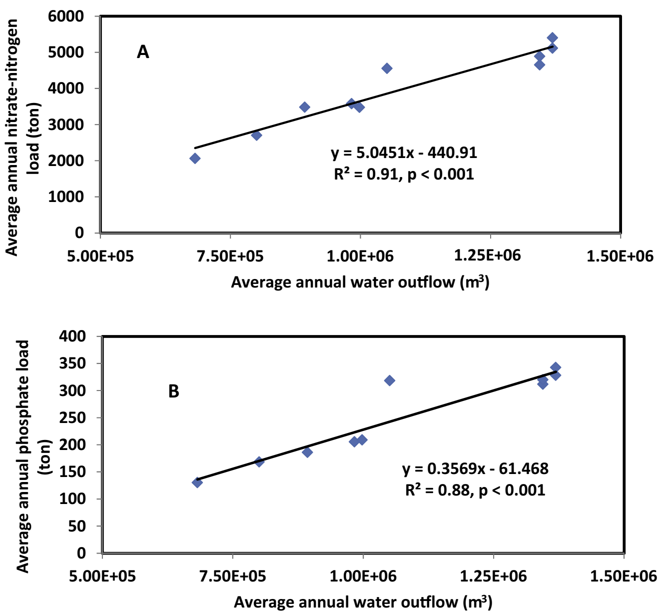

Fig. 5. Relationships of nitrate–N (A) and orthophosphate (B) loads to water outflow from the base simulation scenario.

previous simulations (Ouyang et al., 2013). Fig. 4 shows that effects of rainfall event on daily variations of NO_3^-– – –N and PO_4^{3-} concentrations were profound and these concentrations increased with rainfall rate although a highest rainfall rate at a given time may not correspond well to the highest NO_3^-– – –N or PO_4^{3-} concentration. For example, the highest rainfall rate was found in early 2001 (Fig. 4A), while the highest NO_3^-– – –N concentration was observed in the spring of 2004 (Fig. 4C) and the highest PO_4^{3-} concentration was detected in winter of 2000 (Fig. 4D). It is apparent that other watershed characteristics such as topography, land use cover, soil types, seasonal soil moisture regime, and initial soil N and P contents also play important roles in daily variations of NO_3^-– – –N and PO_4^{3-} concentrations.

Relationships of annual average water outflow to annual NO_3^-– – –N and PO_4^{3-} loads, through the LYRW outlet for the base scenario over a simulation period from 2000 to 2009, are shown in Fig. 5. This figure shows two linear regression equations with very good correlations between water outflow and NO_3^-– – –N load as well as between water outflow and PO_4^{3-} load. With the values of R^2 equal to 0.9053 for NO_3^-– – –N and 0.8455 for PO_4^{3-}, one could use these equations to approximate the annual NO_3^-– – –N and PO_4^{3-} loads for the LYRW given the annual water outflow or vice versa, assuming that the NO_3^-– – –N and PO_4^{3-} concentrations are not reduced naturally or as a result of best management practices. These equations show that the ratio of the average annual water outflow to the average annual NO_3^-– – –N load was 4.567 and to the average annual PO_4^{3-} load was 0.2903. In other words, every 1.0 acre-ft (or 1233.5 m^3) water outflow could bring about 4.567 (metric) ton of NO_3^-– – –N and 0.2903 ton of PO_4^{3-} out of the watershed outlet. It further reveals that the annual load of NO_3^-– – –N was 15.73 (i.e., 4.567 ÷ 0.2903) times higher than the annual PO_4^{3-} load. This was so because the concentration of NO_3^-– – –N in the stream from the LYRW was about 16.74 times higher than the concentration of PO_4^{3-} in the same stream based on the average measured NO_3^-– – –N and PO_4^{3-} concentrations at this watershed.

3.2. Nitrate–N and phosphate load reduction with reforestation

Table 3 compares the seasonal and annual loads of NO_3^-– – –N out of the LYRW outlet among five different levels of land use conversion (i.e., 0, 25, 50, 75, and 100% conversions of agricultural lands into forests) and the corresponding percentage changes of NO_3^-– – –N load reduction. In general, a conversion of agricultural land into forests decreased the seasonal load of NO_3^-– – –N. For example, a 50% reduction of agricultural land (or a 4250 ha increase in forest land (Table 2)) reduced the NO_3^-– – –N load by 4.32% in winter, 9.16% in spring, 6.75% in summer, and 3.86% in fall (Table 3) while a 75% reduction of agricultural land (or a 6375 ha increase in forest land (Table 2)) reduced the NO_3^-– – –N load by 11.87% in winter, 13.42% in spring, 7.57% in summer, and 10.04% in fall (Table 3). Results demonstrated that an increase in forests near or in the batture of the streams reduced NO_3^-– – –N load. This occurred because forest soils enriched in organic matter absorb water and N species and reduce the surface water runoff. Table 3 also shows that the maximum NO_3^-– – –N load reduction with reforestation occurred in spring. This could occur because more fertilizers were applied to the soil during the spring crop planting season. Analogous to the case of average seasonal NO_3^-– – –N load, the average annual NO_3^-– – –N load reduction with reforestation was also observed (Table 3). For instance, a 25% reduction of agricultural land or a 2125 ha increase in forest land (Table 2) reduced the mass of annual NO_3^-– – –N load by 3.02% (Table 3), whereas a 50% reduction of agricultural land or a 4250 ha increase in forest land (Table 2) decreased the mass of annual NO_3^-– – –N load by 6.07% (Table 3). In other words, a two-fold increase in forest land would result in an approximately

Table 3
Comparison of nitrate–N and phosphate load reductions among five percentages of land use conversions from agricultural to forest land.

Time	0% conversion from agricultural to forest land		25% conversion from agricultural to forest land		50% conversion from agricultural to forest land		75% conversion from agricultural to forest land		100% conversion from agricultural to forest land	
NO_3^-– – –N load										
	Load Ton	Change (%)	Load Ton	Change (%)	Load Ton	Change (%)	Load Ton	Change (%)	Load Ton	Change (%)
Winter	1077	0	1053	−2.19	1030	4.32	949	1.87	979	−9.11
Spring	1114	0	1063	−4.57	1012	−9.16	964	−13.42	904	−18.79
Summer	834	0	806	−3.39	777	−6.75	771	−7.57	721	−13.55
Fall	966	0	948	−1.85	929	−3.86	869	−10.04	888	−8.10
Annual	3991	0	3870	−.02	3749	6.07	3553	-10.96	3492	−12.50
PO_4^{3-} load										
Winter	79	0	76	−3.58	73	−7.12	64	−18.98	68	−14.18
Spring	57	0	55	−3.70	53	−7.35	51	−9.84	48	−14.80
Summer	47	0	46	−3.82	44	−7.53	45	−4.96	40	−14.59
Fall	69	0	67	−3.43	64	−7.07	59	−15.37	59	−14.01
Annual	252	0	243	−3.61	234	−7.23	219	−13.30	216	−14.35

two-fold decrease in the mass of annual $NO_3^- - - -N$ load. Results further confirmed that reforestation in or near the batture of the streams is a useful practice for $NO_3^- - - -N$ load reduction. It may be vague to use the percentages (relative values) to compare with other watersheds on how well the nutrient load reduction was after reforestation. For a better comparison, the concept of specific nutrient load reduction was introduced in this study. A specific nutrient load reduction is defined here as the mass of nutrient load reduction per hectare increase in forest land per year. This value was obtained by dividing the mass of annual nutrient load with the total area of forest land increment. For example, a specific $NO_3^- - - -N$ load reduction of 0.056 ton/ha/y means that for every ha increase in forest land, the $NO_3^- - - -N$ flow out of its watershed outlet is reduced by 0.023 ton per year.

Table 2 lists the forest land increment and the specific $NO_3^- - - -N$ load reduction among the five different percentage levels of land use conversion. This table shows that a reduction in agricultural land from 0 to 75% associated an increase in forests, in general, enhanced the specific $NO_3^- - - -N$ load reduction. For example, the specific $NO_3^- - - -N$ load reduction was 0.056 ton/ha/y with a 25% reduction of agricultural land, but was 0.069 ton/ha/y with a 75% reduction of agricultural land. This occurred because the reduction in agricultural land associated with the increase in forests near or in the batture of the streams greatly reduced the surface N runoff. On average, over a 10-year simulation, the specific $NO_3^- - - -N$ load reduction was 0.06 ton/ha/y. That is, per each hectare increase in forests, the $NO_3^- - - -N$ load was reduced by 0.06 ton per year. It is also very interesting to note that although the average annual $NO_3^- - - -N$ load always decreased with increasing forest land conversion, the optimal specific $NO_3^- - - -N$ load reduction was observed at a 75% (not 100%) reduction of agricultural land (Table 2) for the simulation conditions used in this study.

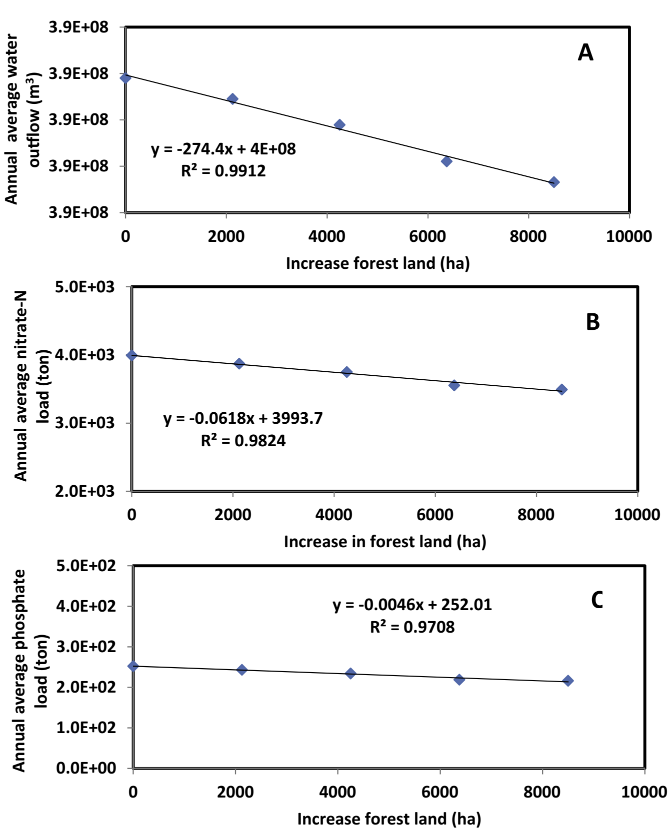

Fig. 6. Relationships of annual average water outflow (A), nitrate–N load (B), and orthophosphate load (C) to forest land increment.

Plot of the annual average water outflow volume against the annual average $NO_3^- - - -N$ load is given in Fig. 5A. With an $R^2 = 0.9053$, we concluded that a highly linear correlation existed between the annual average water outflow volume and the annual average $NO_3^- - - -N$ load. The relationship between the annual average $NO_3^- - - -N$ load and the forest land increment for the LYRW is given in Fig. 6A. With an $R^2 = 0.9824$, we found that an excellent linear correlation existed between the annual $NO_3^- - - -N$ load reduction and the forest land increment. In other words, an increase in forest land area decreased the annual average water outflow (Ouyang et al., 2013) and thereby decreased the annual average $NO_3^- - - -N$ load. Under the assumption that other conditions remained the same except for reforestation, as used in this study, we have demonstrated that the percentage increase in forest land was proportional to the percentage decrease in total mass of $NO_3^- - - -N$ load reduction through the watershed outlet.

Seasonal variations of PO_4^{3-} load reduction from the LYRW outlet among the five different land use conversion rates (i.e., 0, 25, 50, 75, and 100% conversions of agricultural lands into forests) are given in Table 3. A decrease in agricultural land associated with an increase in forest land increased the seasonal PO_4^{3-} load reduction. For instance, a 25% reduction of agricultural land decreased the PO_4^{3-} load through the watershed outlet by 3.58% in winter, 3.70% in spring, 3.82% in summer, and 3.43% in fall; while a 50% reduction of agricultural land reduced the PO_4^{3-} load through the watershed outlet by 7.12% in winter, 7.35% in spring, 7.53% in summer, and 7.07% in fall. Results showed that an increase in forest land near or in the batture of the streams reduced the seasonal PO_4^{3-} load, which occurred because the increase in forest land attenuated the surface water runoff and soil erosion and thereby reducing the PO_4^{3-} load in the streams.

Similar to the case of annual average $NO_3^- - - -N$ load reduction, the annual average PO_4^{3-} load reduction with reforestation was also found among the five different percentages of land use conversion (Table 3). A 25% reduction in agricultural land reduced the annual average PO_4^{3-} load by 3.61%, whereas a 50% reduction in agricultural land decreased the annual average PO_4^{3-} load by 7.07%. A two-fold percentage increase in agricultural land conversion resulted in about a two-fold PO_4^{3-} load reduction. Results further confirmed that reforestation in or near the batture of the streams had a profound reduction in nutrient load.

Table 2 shows the forest land increment and the specific PO_4^{3-} load reduction among the five different percentages of land use conversion for the LYRW. In general, a reduction in agricultural land associated with an increase in forest land enhanced the specific PO_4^{3-} load reduction. For example, the specific PO_4^{3-} load reduction was 0.0043 ton/ha/y with a 25% reduction of agricultural land, but was 0.0053 ton/ha/y with a 75% reduction of agricultural land. On average, over a 10-year simulation, the specific PO_4^{3-} load reduction was 0.004 ton/ha/y. That is, per each hectare increase in forests, the PO_4^{3-} load was reduced by 0.0018 ton per year. These load reductions occurred because of the decrease in agricultural land and concomitant increase in forest land near the batture of the streams that greatly reduced the surface water runoff and soil erosion and thereby enhancing the PO_4^{3-} load reduction.

Fig. 5B shows the relationship between the annual average water outflow volume and the annual average PO_4^{3-} load. With an $R^2 = 0.8455$, we concluded that a highly linear correlation existed between the annual average water outflow volume and the annual average PO_4^{3-} load. The relationship between the annual average PO_4^{3-} load and the forest land increment for the LYRW is given in

Fig. 6B. With an $R^2 = 0.9708$, we concluded that an excellent linear correlation existed between the annual PO_4^{3-} load reduction and the forest land increment. It is apparent that an increase in forest land area decreased the annual average water outflow (Ouyang et al., 2013) and thereby decreasing the annual average PO_4^{3-} load. When other conditions remained the same except for reforestation, a percentage increase in forest land was proportional to the percentage decrease in total mass of PO_4^{3-} load reduction through the watershed outlet.

Mitsch et al. (2001) reviewed the most likely methods for reducing N loading into the GOM from the MRB. These authors concluded that a reduction of about 40% N load into the GOM is possible through the implementation of a number of proven techniques, including reforestation, restoration of wetlands, and riparian buffers. Duffy et al. (1978) studied the impacts of storm water from five reforested watersheds (1.5–2.8 ha) in northern Mississippi during 1974 water year (October 1973–September 1974) on P and sediments. These authors found that the mean concentration of TP in solution is 0.027 mg/L for the five watersheds. Of which, 45% is hydrolyzable P, 33% is ortho-P, and 22% is organic P. These findings further confirmed our conclusion that reforestation reduced N and P loads into streams.

4. Conclusions

Simulations showed that a conversion of agricultural land into forests around or in the batture of the streams greatly reduced $NO_3^- - - -N$ and PO_4^{3-} loads, which occurred because forest soils (enriched in organic matter) absorb water and nutrients, reduce surface runoff, and prevent soil erosion. Overall, a two-fold increase in forest land would result in an approximately two-fold decrease in mass of annual $NO_3^- - - -N$ and PO_4^{3-} loads. Results demonstrated that reforestation in or around the batture of the streams had profound impacts and is a useful practice for nutrient load reduction.

It is also very interesting to note that although the average annual $NO_3^- - - -N$ load decreased with increasing forest land conversion, the optimal specific $NO_3^- - - -N$ load reduction was observed at a 75% (not 100%) reduction of agricultural land for the simulation conditions used in this study.

Under the assumption that other conditions remained the same except for reforestation, we have demonstrated that the percentage increase in forest land was proportional to the percentage decrease in total mass of $NO_3^- - - -N$ and PO_4^{3-} load reduction through the watershed outlet.

The model can be improved by collecting and adding more extensive $NO_3^- - - -N$ and PO_4^{3-} data. Currently, long-term water quality data collection in the LYRB is insufficient to facilitate needed modeling to address specific management and research questions. Therefore, we recommend the initiation of a surface water quality monitoring program for this purpose.

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