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Using HSPF to Model the Hydrologic and Water Quality Impacts of Riparian Land-Use Change in a Small Watershed

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ABSTRACT. Although land-use change in the riparian corridor can be a viable option in mitigating nonpoint source pollution, its impacts under different geographical scales have yet to be ascertained. The goal of this research was to quantify the hydrologic impacts of land-use change in the riparian zones of a subwatershed through the use of an integrated modeling approach. The Hydrological Simulation Program-Fortran model was adopted to develop a hydrologic and water quality model for the Upper Little Miami River basin, a headwater subwatershed in Ohio, USA. After calibration and validation, the model was used to predict the hydrologic and water quality impacts under various scenarios of buffer zones. Results indicated that the 60 m, 90 m, and 120 m riparian forest and wetland buffers were able to reduce the mean annual flow by 0.26 to 0.28%, nitrite plus nitrate by 2.9 to 6.1%, and total phosphorus by 3.2 to 7.8%. Wilcoxon signed rank test for paired data revealed significant differences between the base case (no change in land-use pattern) and scenarios of forest or wetland buffer zones, between pairs of different buffer widths, and between pairs of forest and wetland buffers within a single width level. By integrating environmental information and systems analysis, this study has demonstrated that HSPF is an effective tool to model nonpoint source pollution from riparian land-use changes, even in a small subwatershed with relatively minimal anthropogenic influences. The findings from this research may be useful in facilitating the development of management solutions.

Keywords: watershed modeling, hydrology, water quality, riparian land-use change, HSPF, BASINS

1. Introduction

Most streams and lakes in the United States are affected by nonpoint source (NPS) pollution, such as nutrients, pesticides, and sediments from farms and urban areas (Mitsch et al., 2001). NPS pollution is difficult to control because its sources cannot be attributed to a single particular discharge location. In order to mitigate NPS pollution, we require best management practices (BMPs) and a change in agricultural practices. However, these methods when used alone may not solve all of the nutrient and sediment problems in the surface waters (Vought et al., 1995). A common method to curb pollution is through the establishment of riparian buffers along those areas of stream channels that would be most susceptible to the threat (Narumalani et al., 1997). By changing land-use patterns in these riparian zones, it may help to ameliorate some of these problems. This can be achieved at two scales, either via catchment-wide restrictions or local protection zones (Burt and Johnes, 1997).

The important role of riparian buffer zones in decreasing stream flow and NPS pollution, especially nitrogen (N), phosphorus (P), and sediments, has long been recognized internationally, documented, and used in management practices (Lowrance et al., 1984; Haycock and Burt, 1993; Naiman and Decamps, 1997; Blanco-Canqui et al., 2004; Vidon and Dosskey, 2008; and Diebel et al., 2009). For instance, Lowrance et al. (1984) measured a total P retention of 30% in riparian areas of a watershed in Georgia. Haycock and Burt (1993) indicated that nitrate concentrations in groundwater were reduced by 84% along a grass buffer strip in Cotswolds, England. Blanco-Canqui et al. (2004) reported that runoff was reduced by more than 34.3% with a 0.7 m switchgrass barrier and a 7.3 m vegetative filter strips in Columbia, Missouri. As suggested by Schultz et al. (1995) and Kuusemets and Mander (1999), one of the most effective multi-functional mitigation methods for watershed management is the creation of buffer zones and buffer strips; it is more cost-effective than the conventional conservation or engineering designs.

To assess the effectiveness of riparian buffers in pollution control, one needs to predict the water quality in river catchments with different riparian land-use patterns. According to Lin et al. (2002), the most commonly used approaches in vegetated buffer research are: (1) plot observation and analysis, (2) mathematical evaluation, and (3) mathematical

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modeling combined with the application of geographic information systems (GIS) technology. The mathematical models that incorporate GIS and consider different weather conditions, soil types, slopes, and catchment sizes are very useful as they can help in identifying the critical locations for buffer zones. This modeling approach can be extended to examine, for example, 'what-if' scenarios for various buffer zone widths and conditions (Cryer et al., 2001). Simulation model data can also be used as a surrogate to replace the expensive field data in environmental analyses (Edwards et al., 1996). Thus, an important research agenda is to employ such a modeling approach to analyze the hydrologic and water quality impacts of changing riparian buffer zones (Krysanova et al., 1998; Hattermann et al., 2006; Liu, 2006; Gorsevski et al., 2008).

Currently, many models have been employed to simulate the effectiveness of riparian buffer zones and their impacts on surface runoff from fields. To cite a few examples, Phillips (1989a; 1989b) derived two equations to describe buffer performance, both of which compared a given buffer with a reference buffer. However, as noted by Muscutt et al. (1993), Phillips did not verify his models experimentally, nor were these models calibrated. Lee et al. (1989) developed a mathematical model, GRAPH, to analyze the runoff and phosphorus transport in a single storm using grass buffer strips. Hayes and Dillaha (1992) used the WEPP and GRASSF models to evaluate the effects of grass buffer strips on sediment loads. Hattermann et al. (2006) and Liu et al. (2008) extended the SWIM (Soil and Water Integrated Model) model from SWAT (Arnold et al., 1994) to simulate the water and nutrient flows and retention processes in riparian zones and wetlands; nonetheless, its validation is often difficult to implement due to the scarcity of field data. Gorsevski et al. (2008) integrated the SMR (Soil Moisture Routing) model with probabilistic analysis to identify riparian buffer widths for more effective buffer design. But the SMR model requires variables that are difficult to obtain, such as soil depth, soil porosity, and hydraulic conductivity. The CREAMS (Chemicals, Runoff, and Erosion from Agricultural Management Systems) model (Knisel, 1980) was emp- loyed by Flanagan et al. (1989) and Williams and Nicks (1988, 1993) to examine the efficacy of vegetated buffer strips. Hamlett and Epp (1994) also used the CREAMS model, and they reported that with the 15 m wide grassed strips, surface runoff, sediments and total phosphorus were reduced by about 25, 70, and 80%, respectively. However, CREAMS is not capable of simulating in-stream processes. A site-scale model, REMM (Riparian Ecosystem Management Model), has been developed to couple surface and groundwater filtration functions of buffers (Lowrance et al., 1998; 2000), but the model remains to be validated. By using the REMM model, Chen (2003) documented that the reduced mean stream flow ranged from 0.48 (0.07 m^3/s) to 8.97% (1.27 m^3/s) when the corresponding upland contributing area was increased from 28 to 742 km^2, and the reduced N discharge ranged from 2.4 (4.8 kg/ha) to 17.1% (32.9 kg/ha) when the upland area was increased from 56 to 742 km^2. The SPANS GIS-based simulation study by Perry et al. (1999) also showed that the 30 m buffer reforestation would decrease N and P loadings of 2.7 to 13.2% and 1.5 to 7.4%, respectively. Since a wide range of values was reported in these studies, the impacts of riparian buffer zones are not definitive. There are still major questions on what optimal widths of riparian buffer zones are needed to provide a specific nutrient and sediment load reduction (Osborne and Kovacic, 1993; USEPA, 2005). Besides, most of these studies were focused on large watersheds. More work is therefore needed in quantifying the hydrologic and water quality impacts of riparian land-use change, particularly in a small watershed. Furthermore, the complex intrinsic relationships of riparian land use, water yields, and water quality in different geographical areas are yet to be elucidated. As emphasized by Vaché et al. (2002), research 'based on a wider selection of watersheds in other physiographic and agricultural settings is needed to understand the extent, costs, and benefits of various practices aimed at improving water quality in agricultural regions and downstream.'

To address these research needs, this paper intended to quantify the hydrologic impacts of land-use changes in riparian buffer zones using the Hydrological Simulation Program-FORTRAN, HSPF (Johanson et al., 1980; Donigian et al., 1984; Bicknell et al., 2001), which is incorporated in the Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) package (USEPA, 2001). Currently, HSPF is one of the most comprehensive watershed models that can simulate both urban and agricultural land use, surface and subsurface processes, runoff, sediments, fate and transport of nutrients and other water quality constituents (Donigian et al., 1984; Bicknell et al., 2001). It has been used as the premier nonpoint source model not only by the U.S. Environmental Protection Agency (USEPA), but also by the U.S. Geological Survey (USGS) and the U.S. Army Corps of Engineers in flood forecasting, river basin planning, water quality modeling, as well as assessment of BMPs, climate, and land-use changes (Albek et al., 2004; Ackerman et al., 2005; Mishra et al., 2007; Ribarova et al., 2008; and Topalova et al., 2009). The Upper Little Miami River (Upper LMR) basin, a small headwater subwatershed in southwest Ohio, USA, was chosen as a case study. The specific objectives for this first HSPF-based riparian land-use change research were: (1) to test the efficacy of HSPF in modeling hydrologic changes in a small subwatershed, (2) to quantify the effects of riparian land-use types on water quantity and quality (nutrientts: N and P) using various land-use change scenarios, and (3) to compare the effects of different widths of riparian buffer zones on water quality and quantity.

2. Methodology

2.1. Study Area

The Little Miami River (LMR) is a major tributary of the Ohio River in southwest Ohio, USA. It flows southwesterly and joins the Ohio River near the Greater Metropolitan Cincinnati area. The watershed occupies 4,550 km^2 of land area and encompasses eleven counties. LMR was once a pristine river system with minimal anthropogenic impacts. Various sections of its mainstem have been designated as scenic rivers since 1969. However, recently, the point and nonpoint source pollutants as well as habitat alterations from agricultural and

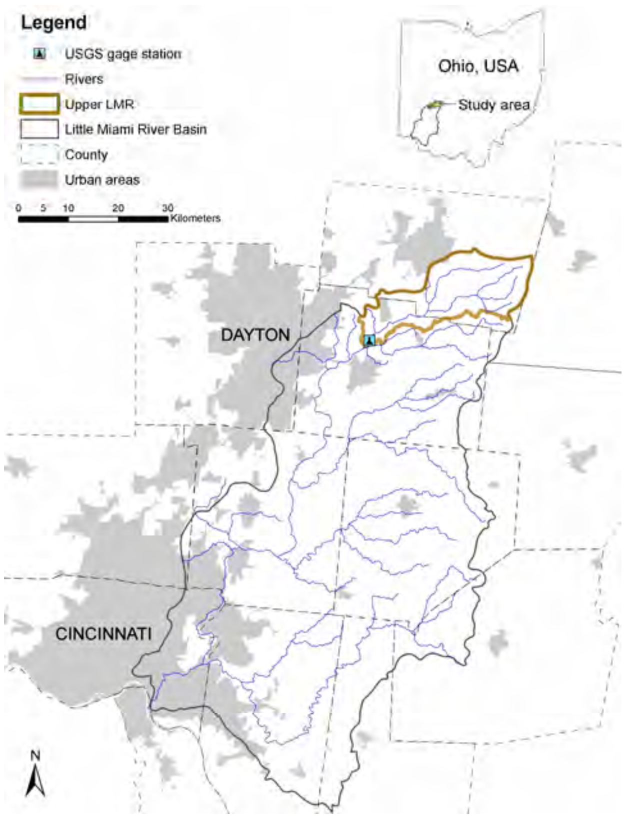

Figure 1. Locations of the Little Miami River and the Upper Little Miami River watersheds, Ohio.

urban land uses have impacted the streams throughout the watershed. For example, it was found that in the 1990s, the median concentration of total phosphorus in the mainstem was 0.34 mg/L (Ohio EPA, 2000), which was greater than the minimum detection limit of 0.05 mg/L. With the impending urbanization and suburban sprawl, the effects of land-use changes on water quality are becoming a priority for scientific research in the study area; see for example, Tong (1990), Wang (2001), and Tong and Naramngam (2007). Nevertheless, none of these studies on the LMR had examined the specific impacts of riparian land use on water quality, although changing land-use patterns in the riparian zones, rather than in the whole watershed, is more feasible and economical.

In this study, the 334 km^2 subwatershed of the LMR headwater near Old Town (the Upper LMR watershed, see Figure 1) was selected for detailed analyses. This is the uppermost 11-digit hydrologic unit in the LMR basin, which covers the Clark and Greene Counties. Based on the 1990s land-use map, the study area consists of 87% agriculture, 9% forest, 4% urban, and some water body (<1%).

2.2. Data

The map of the 11-digit hydrologic units for the State of Ohio was obtained from the USGS, and it was used as the base map for the analyses. Core data were available from the USEPA Region 5 dataset and were extracted from the BASINS package. These data included a 1980s land-use map from the U.S. Department of Agriculture (USDA), a soil layer at a 30 m × 30 m cell size from the USDA National State Soil Geographic Data Base (STATSGO), National Hydrography Dataset (NHD) coverages from the USEPA, and a digital elevation model (DEM, 30 m × 30 m resolution) from the USGS. The climate data were extracted from the weather data management (WDM) files in the BASINS package. WDM files store weather data, such as hourly and daily precipitation, evaporation, temperature, wind speed, solar radiation, potential evaporation, dew point temperature, and cloud cover, from different meteorological stations. Daily meteorological values from the Dayton Airport station in the WDM files were chosen in this study. This meteorological station is the closest station to the study area, and it has the longest and most complete data. The 1990s land-use coverage (Figure 2a) was obtained from the Multi-Resolution Land Characteristics Consortium (MRLC, 30 m cell size).

The Upper LMR basin was delineated with the BASINS manual delineation tool using the NHD and DEM data layers. The USGS gauging station near Old Town (gauging number: 03240000) was selected as the outlet of the watershed (Figure 1). The historical flow and water quality data from 1978 to1993, used in model calibration and validation, were obtained from the USGS (NWIS, http://waterdata.usgs.gov/nwis) and USEPA (STORET, http://www.epa.gov/storet) websites. These data were geo-referenced and geo-processed in ArcView GIS version 3.2 (ESRI Inc., Redlands, CA, USA). They were resized, clipped, merged, and projected to the same coordinate system to ensure conformity with each other.

In this research, the variables flow, nitrite plus nitrate (NO_2 + NO_3), and total phosphorus (TP) were used to denote the overall water quantity and quality conditions. This is mainly because changes in buffer zones often affect these variables in the receiving water bodies (Norris, 1993). Moreover, the literature on LMR has indicated that these variables are related to land-use changes in the watershed (Wang and Yin, 1997; Liu et al., 2000; Tong and Chen, 2002; Tong et al., 2008). The selected study time period was the water years 1980~1993 (10/1/1979 to 9/30/1993), generally consistent with the land-use data (1980s and 1990s).

2.3. HSPF

In this research, the HSPF model was run from BASINS, which is a GIS-based multipurpose environmental analysis system, integrating environmental data, analytical tools, and modeling programs to support the development of cost-effective approaches to environmental protection (USEPA, 2001; 2004). HSPF was first developed as the Stanford Watershed Model (Crawford and Linsley, 1966). It is a semi-distributed conceptual model that combines spatially distributed physical attributes into hydrologic response units, each of which is assumed to behave in a uniform manner (Johnson et al., 2003). HSPF consists of three basic application components: PERLND (Pervious Land Segment), IMPLND (Impervious Land Segment), and RCHRES (free-flowing reach or

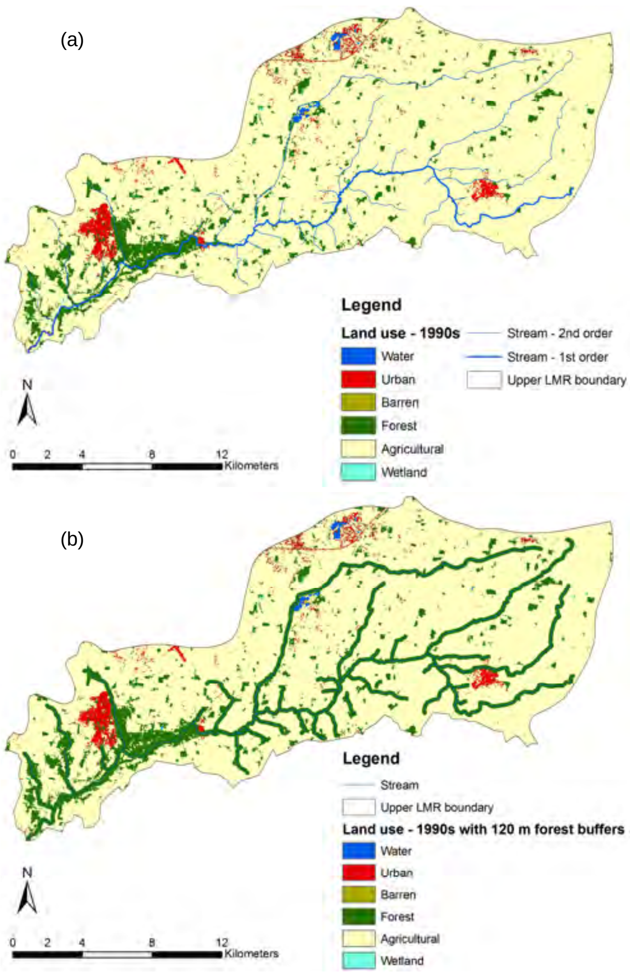

Figure 2. Land-use map of the Upper Little Miami River Basin, 1990s. (a) Original land use overlaid with 1st and 2nd order streams; (b) Land-use scenario with the 120 m riparian forest buffers for the 1st and 2nd order streams.

mixed reservoirs). In HSPF, the various hydrologic processes are represented mathematically as flows and storages. In general, each flow is an outflow from a storage and is usually expressed as a function of the current storage amount and the physical characteristics of the subsystem. HSPF relies on the use of calibrated parameters, and it does not require specific information on the physical dimensions and characteristics of the flow system. This reduces input requirements and enhances the generality of the model (Bicknell et al., 2001).

In HSPF, N and P are modeled using simple empirical relationships in the nutrient cycles under the modules of PQUAL (quality constituents for pervious land), IQUAL (quality constituents for impervious land), and GQUAL (general quality constituents for reaches) for PERLND, IMPLND, and RCHRES, respectively. Besides, more comprehensive modules in the Agrochemical Sections and RQUAL (constituents involved in biochemical transformations) for PERLND and RCHRES are available. As nitrites and nitrates are generally not adsorbed to the sediments, the $NO_2 + NO_3$ simulation uses the method of accumulation and depletion/removal with a first-order washoff rate. In the case of TP, the phosphorus load is often correlated to the sediment load. Hence, the modeling of TP uses a potency factor approach that takes into account the relationship of the strength of the constituent with sediment removal. The N and P for each flow path are estimated on a unit area basis for each land use. These are then added to the rivers and routed to the basin outlet. Chemical, biological, and physical in-stream processes are also simulated.

Table 1. General Calibration/Validation Targets* for HSPF Applications

Type	Very good	Good	Fair
Hydrology/flow	< 10	10-15	15-25
Sediment	< 20	20-30	30-45
Water temperature	< 7	8-12	13-18
Water quality/nutrients	< 15	15-25	25-35
Pesticides/toxics	< 20	20-30	30-40

*Shown in % difference between simulated and recorded values;
% difference = [(Simulated - Observed) / Observed] ×100;
Source: Donigian, 2000.

2.4. Model Calibration and Validation

After the model for the study area is developed, calibrtion and validation are needed to ascertain its accuracy and robustness. As suggested by Donigian (2002), the overall performance of the model can be evaluated through graphical comparisons and statistical tests. Table 1 lists a general guideline for HSPF evaluation in terms of the percent mean errors or differences between simulated and observed values.

In this study, the hydrologic and water quality calibrations were carried out for the water years 1980 to 1986 (10/1/1979 to 9/30/1986), and the validation was performed using the data from the water years 1987 to 1993 (10/1/1986 to 9/30/1993). Each period included several dry (low flow) years and wet (high flow) years. This research used two water years, 1978 and 1979, as a 'spin-up' period to ensure a better representation of the initial watershed conditions. Pearson correlation coefficient (r, range: -1.0 to 1.0) was used to evaluate the agreement between observed and simulated flow and nutrient data. The Nash-Sutcliffe model efficiency coefficient (E, range: $-\infty$ to 1.0, Nash and Sutcliffe, 1970) was calculated to assess the predictive power of hydrologic models, indicating how consistently observed values would match predicted values. Additionally, we computed the coefficient of determination (R^2, the square of Pearson correlation coefficient, r) and root-mean-square error (RMSE), both of which are widely used measures to test for the goodness-of-fit from hydrologic modeling results (Kim et al., 2007).

2.4.1. Calibration and Validation of the Hydrologic Model

A hydrologic model for the Upper LMR basin was built by first running HSPF using the default parameters. Average sampled daily flow discharges from four main pollution point sources in the study area were added into HSPF using the point

Table 2. HSPF Parameters Adjusted for Hydrologic Calibration and Validation

Parameter*	Unit**	Default value	Modified value	Typical range	Possible range	Singh et al. (2005) ***	El-Kaddah and Carey (2004) ***
LZSN	in	6	2	3.0-8.0	2.0-15.0	5	15
INFILT	in/hr	0.16	0.065	0.01-0.25	0.001-0.5	0.2	varies
AGWRC	1/day	0.98	0.99	0.92-0.99	0.85-0.999	0.98	0.98
DEEPFR	-	0.1	0.4	0-0.2	0-0.5	0.05	0.1
UZSN	in	1.128	1	0.1-1.0	0.05-2.0	0.2-1.4	2.5
INTFW	-	0.75	3	1.0-3.0	1.0-10.0	1.2-1.8	0.75
LZETP	-	0.1	0.7	0.2-0.7	0.1-0.9	0.1-0.75	0.1-0.8

*LZSN is lower zone nominal soil moisture storage, which is within the possible range. INFILT is index to infiltration capacity. AGWRC is base groundwater recession. DEEPFR is fraction of GW inflow to deep recharge, which is within the possible range. UZSN is upper zone nominal soil moisture storage. INTFW is interflow inflow parameter. LZETP is lower zone ET parameter. Typical and possible ranges are from BASINS Technical Note 6 (USEPA, 2000).
**English units are originally listed in the BASINS Technical Notes (1 in = 2.54 cm).
***Parameter values used by other researchers.

source editor. Through trial-and-error, the model parameters were adjusted, and the hydrologic model was calibrated (Table 2). A computer program, HSPEXP (Lumb et al., 1994), was also used to facilitate the calibration.

2.4.2. Calibration and Validation of the Water Quality Model

After the hydrologic model was calibrated and validated, two water quality models, nitrite plus nitrate as nitrogen (NO_2 + NO_3) and total phosphorus (TP), were created. The point source discharge information in the study area was retrieved from the BASINS database and added to the model. Various parameter values were adjusted for each land-use category and for each month to account for seasonal factors. After the simulation, the model was calibrated by comparing the simulation results with the historical water quality data from the USGS and USEPA. Since there are no specific guidelines for water quality calibration in HSPF (Donigian, 2002), researchers, such as El-Kaddah and Carey (2004) and Singh et al. (2005), often adjust the parameters values in a trial-and-error fashion until the simulated concentrations match relatively closely to the observed ones, while maintaining the parameters within physically realistic bounds. Based on literature and BASINS technical guides (USEPA, 2000; El-Kaddah and Carey, 2004; Tong and Liu, 2006), several parameters were first selected for calibration (Table 3). After numerous trial-and-error model runs, it was noticed that two water quality parameters, Mon-IFLW-CONC and Mon-GRND-CONC, had significant impacts on the nitrogen and phosphorus yields in the study area. Mon-IFLW-CONC (mg/L) is the monthly nutrient concentration in interflow. Mon-GRND-CONC (mg/L) refers to monthly nutrient concentration in groundwater. The other three parameters, SQOLIM (the maximum amount of the pollutant that can be stored on the land surface), WSQOP (the rate at which the pollutants are washed off the surface), and POTFS (the ratio of constituent yield to sediment outflow) had slight to moderate impacts on the nutrients yield.

2.5. Riparian Land-Use Scenarios

In this study, riparian land-use scenarios were generated by changing current land uses in the riparian buffers into either forest or wetland in ArcView GIS. Since the resolution of the 1990s MRLC land-use data is 30 m, the minimum buffer width has to be at least 30 m. In addition, because HSPF land segments are modeled using a unit land area (1 acre, or about 4,047 m^2), the minimum buffer width of 60 m was needed. Hence, three different (60, 90, and 120 m) buffer zones along the first and second order reaches/streams (Figure 2a) were created in GIS. Of the total reaches in the subwatershed, the cumulative percentage of these two types of reaches was 56.3% (16.2 and 40.1% for the first and second orders, respectively). Based on the 1990s MRLC land-use data, which were used as the base case (no change in land use) scenario, these land uses in riparian buffer zones were converted to only either forest or wetland land-use types. These constituted six riparian scenario cases for the simulation. The changes in terms of areas and percentages in each of these land-use types in the riparian buffers were listed in Table 4. Figure 2b showed an example of the 120 m riparian forest buffer scenario.

SAS statistical software (SAS Institute Inc., Cary, NC, USA) was used to modify GIS land-use dbf tables. Using the 'land-use editor' in BASINS, the existing land-use pattern was modified and loaded to the validated model. For each riparian land-use scenario, a separate hydrologic and water quality model was created with validated model parameters. Altogether, there were six models with different riparian land-use scenarios and one base model with no change in the land-use pattern. Using the calibrated/validated parameter values, model simulations (1986 to 1993) on each riparian land-use scenarios were performed in two steps: the simulation of the hydrology (flow) and the simulation of the water quality parameters (NO_2 + NO_3 and TP). Wilcoxon signed rank test for paired data (Siegel and Castellan, 1988), the non-parametric equivalent of the paired *t*-test, was used to examine the statistical differences between selected pairs of scenarios for flow, NO_2 + NO_3, and TP. Altogether, there were 15 paired comparisons: the base case and each of the six forest/wetland scenarios, six pairs of different width levels for forest and wetland, respectively (for example, 60 vs. 90 m forest), and three pairs within a single width level (for example, 60 m forest vs. 60 m wetland). Two-tailed p-values less than 0.05 were considered as statistically significant.

Table 3. HSPF Parameter Values Adjusted for Water Quality Calibration and Validation*

Mon-IFLW-CONC ($NO_2 + NO_3$)**	Water	Urban	Barren	Forest	Agricultural	Wetland
JAN	5 (0.5)	8 (1.5)	6 (1)	5 (0.6)	8 (1)	5 (0.6)
FEB	3 (0.5)	6.3 (1.5)	4.8 (1)	2.8 (0.6)	5.3 (1)	2.8 (0.6)
MAR	2.8 (0.5)	3.3 (1.5)	5.3 (3)	2.8 (0.6)	5.3 (3)	2.8 (0.6)
APR	0.1 (0.3)	0.8 (2.5)	1 (18)	0.1 (0.4)	5 (18)	0.1 (0.4)
MAY	1.2 (0.3)	1.4 (2.5)	6.5 (19)	1.2 (0.4)	6.5 (19)	1.2 (0.4)
JUN	1.2 (0.3)	1.4 (2.5)	6 (15)	1.2 (0.4)	6.5 (15)	1.2 (0.4)
JUL	1.5 (0.3)	2 (2.5)	6.2 (15)	1.6 (0.4)	6.2 (15)	1.6 (0.4)
AUG	0.92 (0.3)	1.4 (2.5)	5 (12)	0.92 (0.4)	5 (12)	0.92 (0.4)
SEP	0.15 (0.3)	1 (2.5)	5 (12)	0.2 (0.4)	5 (12)	0.2 (0.4)
OCT	0.1 (0.7)	0.6 (1.5)	4.5 (12)	0.1 (0.8)	4.5 (12)	0.1 (0.8)
NOV	0.6 (0.7)	1.5	4.9 (5)	0.7 (0.8)	4.9 (5)	0.7 (0.8)
DEC	2.6 (0.7)	5 (1.5)	6.5 (2)	2.8 (0.8)	6.5 (2)	2.8 (0.8)
Mon-GRND-CONC ($NO_2 + NO_3$)	Water	Urban	Barren	Forest	Agricultural	Wetland
JAN	4 (0.5)	7 (1.5)	6.5 (1)	4 (0.5)	6.5 (1)	4 (0.5)
FEB	2.8 (0.5)	6.1 (1.5)	4.3 (1)	2.8 (0.5)	5.3 (1)	2.8 (0.5)
MAR	3 (0.5)	5.8 (1.5)	5.8 (2)	2.8 (0.5)	6.3 (2)	2.8 (0.5)
APR	0.17 (0.3)	0.4 (2.5)	0.7 (12)	0.17 (0.3)	4 (12)	0.17 (0.3)
MAY	0.9 (0.3)	1.3 (2.5)	4.5 (12)	0.9 (0.3)	4 (12)	0.9 (0.3)
JUN	0.9 (0.3)	1.2 (2.5)	2.6 (10)	0.9 (0.3)	3.9 (10)	0.9 (0.3)
JUL	1.4 (0.3)	2 (2.5)	5.2 (10)	1.4 (0.3)	5.2 (10)	1.4 (0.3)
AUG	0.94 (0.3)	1.4 (2.5)	3.3 (7)	0.84 (0.3)	3.1 (7)	0.94 (0.3)
SEP	0.2 (0.3)	2 (2.5)	3 (7)	0.2 (0.3)	3 (7)	0.2 (0.3)
OCT	0.18 (0.7)	1 (1.5)	2 (7)	0.18 (0.7)	2.7 (7)	0.18 (0.7)
NOV	0.6 (0.7)	1.4 (1.5)	3 (4)	0.6 (0.7)	3 (4)	0.6 (0.7)
DEC	2.5 (0.7)	4.8 (1.5)	4.8 (1.5)	2.4 (0.7)	4.8 (1.5)	2.4 (0.7)
Mon-IFLW-CONC (TP)	Water	Urban	Barren	Forest	Agricultural	Wetland
JAN	0.01	0.03 (0.05)	0.08 (0.1)	0.01	0.08 (0.1)	0.01
FEB	0.03 (0.01)	0.08 (0.05)	0.38 (0.1)	0.03 (0.01)	0.38 (0.1)	0.03 (0.01)
MAR	0.03 (0.01)	0.09 (0.05)	0.28 (0.1)	0.03 (0.01)	0.28 (0.1)	0.03 (0.01)
APR	0.05 (0.01)	0.04 (0.05)	0.1 (0.1)	0.05 (0.01)	0.01 (0.1)	0.05 (0.01)
MAY	0 (0.01)	0.01 (0.05)	0.02 (0.1)	0 (0.01)	0.15 (0.1)	0 (0.01)
JUN	0.04 (0.01)	0.25 (0.05)	0.45 (0.1)	0.04 (0.01)	0.45 (0.1)	0.04 (0.01)
JUL	0.02 (0.01)	0.1 (0.05)	0.2 (0.1)	0.02 (0.01)	0.2 (0.1)	0.02 (0.01)
AUG	0.01	0.3 (0.05)	0.08 (0.1)	0.01	0.5 (0.1)	0.01
SEP	0.09 (0.01)	0.35 (0.05)	0.65 (0.1)	0.09 (0.01)	0.75 (0.1)	0.09 (0.01)
OCT	0.02 (0.01)	0.1 (0.05)	0.18 (0.1)	0.02 (0.01)	0.36 (0.1)	0.02 (0.01)
NOV	0.02 (0.01)	0.1 (0.05)	0.22 (0.1)	0.02 (0.01)	0.22 (0.1)	0.02 (0.01)
DEC	0.01	0.07 (0.05)	0.1	0.01	0.13 (0.1)	0.01
Mon-GRND-CONC (TP)	Water	Urban	Barren	Forest	Agricultural	Wetland
JAN	0 (0.01)	0.01 (0.03)	0.02 (0.05)	0 (0.01)	0.02 (0.05)	0.01
FEB	0 (0.01)	0.02 (0.03)	0.04 (0.05)	0 (0.01)	0.03 (0.05)	0.01
MAR	0.01	0.03	0.05	0.01	0.05	0.01
APR	0.01	0.02 (0.03)	0.02 (0.05)	0.01	0.05	0.01
MAY	0 (0.01)	0.01 (0.03)	0.02 (0.05)	0 (0.01)	0.02 (0.05)	0 (0.01)
JUN	0.05 (0.01)	0.03	0.05	0.05 (0.01)	0.05	0.01
JUL	0 (0.01)	0.01 (0.03)	0.03 (0.05)	0 (0.01)	0.03 (0.05)	0 (0.01)
AUG	0.01	0.06 (0.03)	0.08 (0.05)	0.01	0.08 (0.05)	0.01
SEP	0.09 (0.01)	0.02 (0.03)	0.09 (0.05)	0.09 (0.01)	0.09 (0.05)	0.01
OCT	0.01	0.06 (0.03)	0.08 (0.05)	0.01	0.08 (0.05)	0.01
NOV	0.01	0.06 (0.03)	0.08 (0.05)	0.01	0.08 (0.05)	0.01
DEC	0 (0.01)	0.02 (0.03)	0.04 (0.05)	0 (0.01)	0.04 (0.05)	0.01

* Numbers in parenthesis were default values; parameters not listed in the table include: SQOLIM: the maximum storage of QUALOF in section PQUAL of PERLND, equals 0.1 (0). POTFS: scour potency factor, in section PQUAL of PERLND, equals 20 (0). WSQOP: the rate of surface runoff that will remove 90 percent of stored QUALOF per hour (cm/hr), equals 5.5 (0.5) in section PQUAL of PERLND, and equals 2 (0.5) in section IQUAL of IMPLND.

** Mon-IFLW-CONC and Mon-GRND-CONC: Monthly NO_2 + NO_3 or TP concentrations in interflow and groundwater (mg/L), repectively, in section PQUA of PERLND (Pervious Land).

Table 4. Land-Use Distributions in Riparian Buffers (1990s) and Major Land-Use Changes* in Upper LMR Basin (unit: km^2)

Type	Agriculture	Forest	Wetland	Urban	Water
60 m buffer	13.8	4.4	0.4	0.3	0.1
90 m buffer	21.2	5.9	0.5	0.4	0.1
120 m buffer	28.6	7.3	0.5	0.5	0.2
Base case - 1990s	297.1 (88%)	30.7 (9.1%)	1.0 (0.3%)	7.7 (2.3%)	1.0 (0.3%)
60 m forest buffers	-13.8 (84.1%)	+14.1 (13.3%)	-0.4 (0.2%)	-0.3 (2.2%)	-
90 m forest buffers	-21.2 (81.9%)	+21.5 (15.5%)	-0.5 (0.1%)	-0.4 (2.2%)	-
120 m forest buffers	-28.6 (79.7%)	+29.2 (17.8%)	-0.5 (0.1%)	-0.5 (2.1%)	-
60 m wetland buffers	-13.8 (84.1%)	-4.6 (7.8%)	+18.2 (5.7%)	-0.3 (2.2%)	-
90 m wetland buffers	-21.2 (81.9%)	-6.2 (7.3%)	+27.3 (8.4%)	-0.4 (2.2%)	-
120 m wetland buffers	-28.6 (79.7%)	-7.5 (6.9%)	+36.2 (11%)	-0.5 (2.1%)	-

*The data shown in the lower part of the table were changes from the base case scenario. Numbers in parenthesis were percentages over the total land-use area within each simulation scenario. Two other land-use categories, barren and water, had almost the same percentages for all scenarios: barren (0.01%) and water (0.3%).

Table 5. HSPF Model Calibration and Validation Results Comparing Simulated and Observed Values (Mean Daily Values)

Type	Parameter	Observed	Simulated	Percent difference*	r**	r^2	RMSE	E
Calibration (1980-1986)	Water flow (m^3/s)	4.03	3.52	-12.46	0.81	0.66	4.03	0.66
	Nitrite plus nitrate, $NO_2 + NO_3$ (mg/L)	4.325	4.32	0.1	0.77	0.59	0.63	0.52
	Total phosphorus, TP (mg/L)	0.099	0.11	-9.98	0.85	0.72	0.019	0.59
Validation (1987-1993)	Water flow (m^3/s)	3.53	4.05	14.61	0.76	0.58	4.64	0.35
	Nitrite plus nitrate, $NO_2 + NO_3$ (mg/L)	4.314	4.48	-3.71	0.81	0.66	0.56	0.45
	Total phosphorus, TP (mg/L)	0.104	0.103	1.08	0.82	0.67	0.02	0.14

*Percent difference = [(Simulated - Observed) / Observed] ×100.
**r: Pearson correlation coefficient, for water flows, the daily r is reported; for nutrients, monthly r are computed, since available observed water quality concentrations were limited. For $NO_2 + NO_3$, N = 88 (calibration); N = 84 (validation). For TP, N = 86 (calibration); N = 81 (validation).

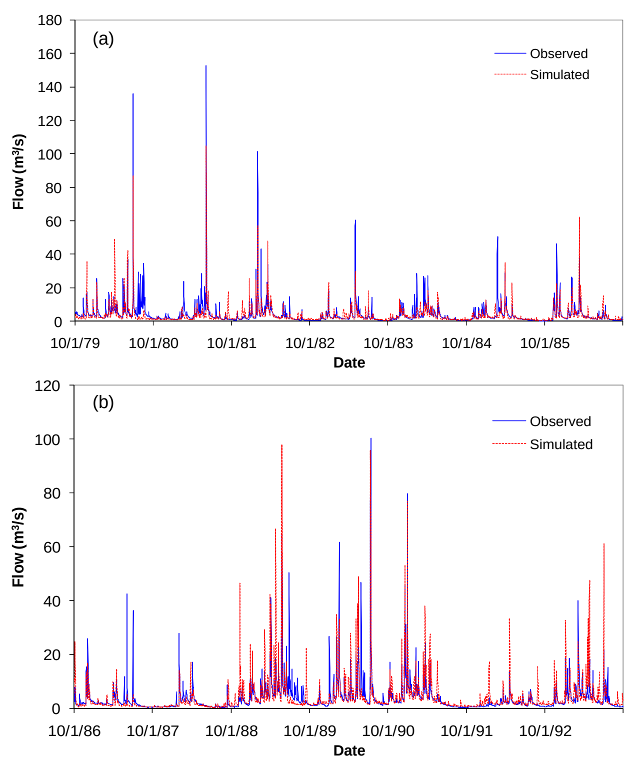

Figure 3. Comparison between observed and simulated daily flow: (a) calibration and (b) validation.

3. Results and Discussion

3.1. Calibration and Validation Results

3.1.1. Hydrologic Model

When compared with the historical mean daily values of flow (Table 5), the results showed good agreement with percent differences of -12.46 % for calibration and 14.61% for validation (Figures 3 to 5), placing it in a 'good' category in terms of HSPF model efficiency targets (Table 1). The average daily correlation coefficients were 0.81 for calibration and 0.76 for validation (Figure 4). The Nash-Sutcliffe efficiency (E) was 0.66 for flow calibration and 0.35 for validation. These results indicated that the hydrologic model could accurately simulate the flow regime. Figure 4 illustrated that the simulation could capture most of the streamflow peaks and seasonal changes. However, the RMSEs were relatively high for both time periods (4.03 and 4.64, respectively), which might be due to some storm peakflows.

The flow-duration curves (Figure 5) are cumulative frequency plots, which show how well the two series correlate in frequency. Overall, the flow-duration curves indicated that the simulations represented streamflow reasonably well. However, for the calibration period, the lowest 20% of flows were slightly over simulated (simulated > observed, Figure 5a). For the validation period (Figure 5b), the lowest 2% of flows were slightly under simulated (simulated < observed), and the high-

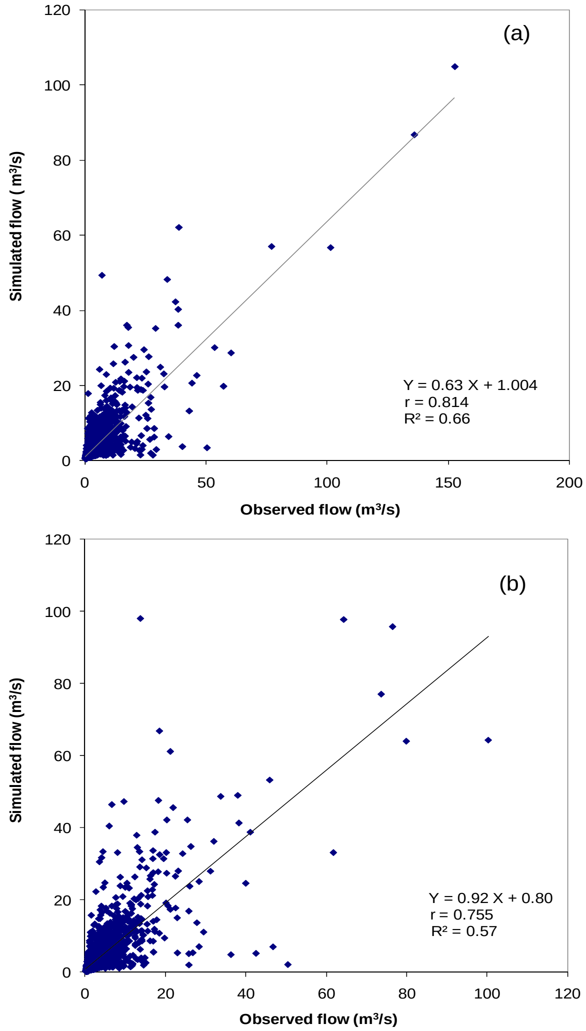

Figure 4. Scatter plots of observed and simulated daily streamflow: (a) calibration and (b) validation.

est 15% of flows were over simulated. Based on these results, the hydrologic model was regarded to be adequate to simulate the flow conditions of the Upper LMR basin.

3.1.2. Water Quality Model

Table 5 and Figures 6 and 7 present the calibration and validation results for the water quality model. In this study, the water quality constituents were simulated on a daily basis, while observed concentrations were only available for limited days. For NO_2 + NO_3 calibration, the simulated and observed mean monthly concentrations were 4.325 and 4.32 mg/L, respectively; for validation, those values were 4.314 and 4.48 mg/L, respectively. These results indicated a good agreement between simulated and observed NO_2 + NO_3 concentrations with percent differences of 0.1% for calibration and -3.71% for validation. According to Table 1, the performance of NO_2 + NO_3 simulation was 'very good'. Moreover, the average monthly correlation coefficients were 0.77 for calibration and 0.81 for validation. The Nash-Sutcliffe efficiency (E) was 0.52

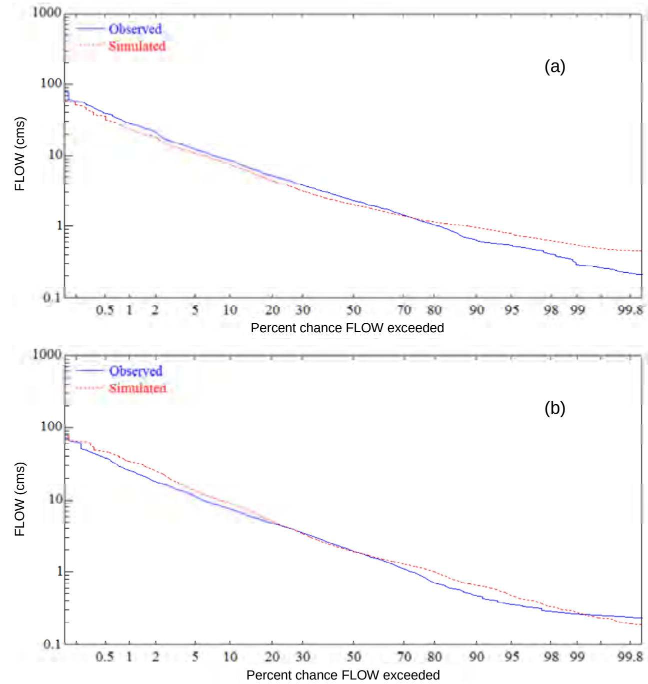

Figure 5. Flow duration curves of simulated and observed daily streamflow: (a) calibration and (b) validation (unit: cubic meters per second).

for calibration and 0.45 for validation.

For the TP calibration, the simulated and observed mean monthly concentrations were 0.099 and 0.11 mg/L, respecttively; for the validation, those values were 0.104 and 0.103 mg/L, respectively. Table 5 demonstrated that a good agreement between simulated and observed TP concentrations was attained with percent differences of -9.98% for calibration and 1.08% for validation. According to Table 1, the TP simulation model performance was also 'very good'. The Nash-Sutcliffe efficiency (*E*) was 0.59 for calibration and 0.14 for validation. The average monthly correlation coefficients were 0.85 for calibration and 0.82 for validation.

Based on the above calibration and validation results, it was concluded that the hydrologic and water quality model in the Upper LMR basin was accurate enough to assess the hydrology and water quality impacts of riparian land-use changes.

3.2. Simulation of the Hydrologic Effects of Riparian Land-Use Changes

It was expected that changing riparian land uses into forest and wetland would decrease flow quite noticeably, as reported in some of the literature (for example, 34% from Hamlett and Epp, 1994; 25% from Blanco-Canqui et al., 2004). However, in this study, the 60 m riparian forest and wetland buffers reduced the mean annual flow only by 0.256 and 0.258%, respectively (Table 6). The decreases from 90 and 120 m riparian forest and wetland buffers were 0.26 and 0.28%, respectively. The mean annual flow reductions ranged from 3.79 to 4.17 m^3/s. The change in the mean daily flow was 0.01 m^3/s for each of the six scenarios. The highest daily reductions

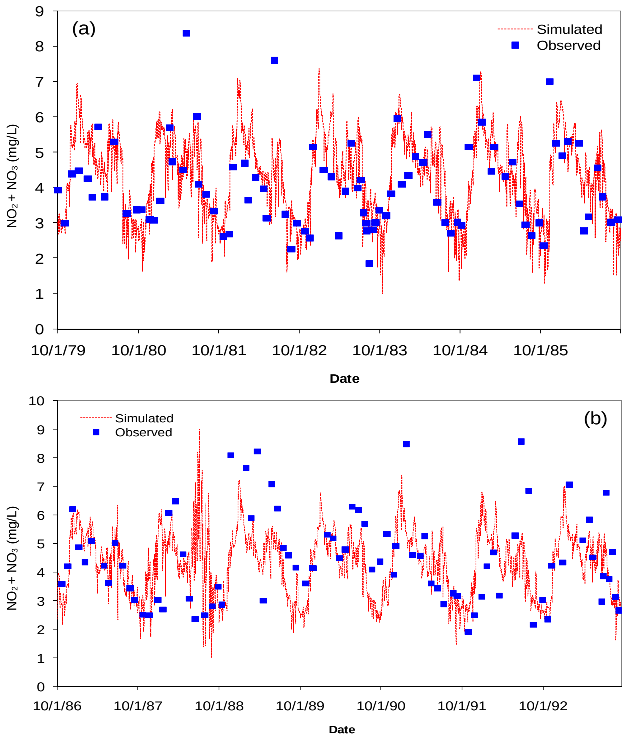

Figure 6. Comparison between simulated and observed $NO_2 + NO_3$ (mg/L): (a) calibration and (b) validation.

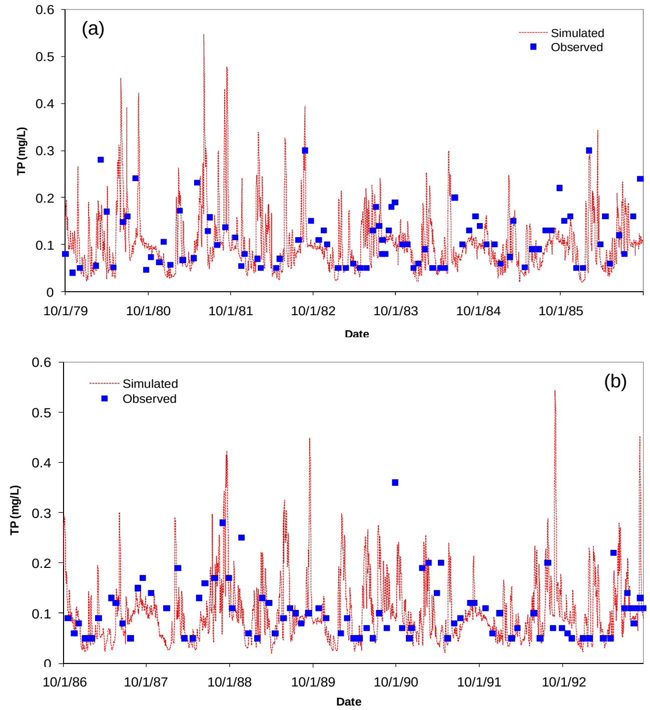

Figure 7. Comparison between simulated and observed TP (mg/L): (a) calibration and (b) validation.

for the 60 and 120 m riparian forest buffers scenarios occurred in July of 1990 (over 0.34 and around 0.42 m^3/s, respectively).

Contrarily to other field-based studies (Blanco-Canqui et al., 2004) and modeling simulations (Hamlett and Epp, 1994), these results indicated that riparian land-use changes for these three width levels and two land-use types had little impacts on flow volume in this headwater stream. The achieved flow reduction was only comparable to the lowest results obtained from the REMM simulations (0.48%, Chen, 2003). There was not much difference among the 60, 90, and 120 m riparian forest buffers. Furthermore, the impacts on flow from forest and wetland buffers were almost the same. However, test results from Wilcoxon signed rank test showed that, except for the comparison between 90 m forest and 90 m wetland buffers, significant differences were identified for all other 14 paired comparisons (d.f. = 2 556; all $p \leq 0.005$).

Several reasons might explain the limited effectiveness of riparian buffer strips in reducing flow in this study area. First, it might be related to the spatial scale and geographical location used in the study. Since HSPF uses the amalgamated percentages of land-use types in each subwatershed, it may not be effective in capturing the hydrologic impacts of land-use changes in each small buffer zone within the subwatershed. In order to improve the simulation results, we might have to separate each individual riparian zone from the subwatershed and model its hydrologic processes independently. Also, in this small watershed study, the impacts from land-use changes around lower orders of rivers were simulated (the first and second orders, with 16.2 and 40.1% of all rivers, respectively). However, in other research, the hydrologic impacts from land-use changes in a much larger watershed and higher order streams were generally modeled. Furthermore, this study was carried out in the headwater streams of the Upper LMR basin. As suggested by Leeds-Harrison et al. (1999), in headwaters, the precipitation quickly turns into surface runoff especially during intensive rainfall. This reduces the flow retention capabilities of riparian forest and wetland buffer zones. Consequently, the impacts of land-use changes in the headwater buffer strips might be smaller. Second, it might be related to the extent and types of land-use changes. From the literature, large flow reductions were often associated with greater percentage changes in land-use patterns. For example, in the REMM study, the upland forested area was increased by more than 714 km^2 (Chen, 2003). However, in this study, the increase in forest land only ranged from 14.1(3,475 acres) to 29.2 km^2 (7,207 acres, Table 3). Riparian buffers were created for only 56% of streams (namely, first and second order rivers). Future research is therefore needed to include the other 44% of higher order rivers in the study area to fully examine the buffering function on stream flow. It also seems that urban land use has more impacts on flow. Studies of urban watersheds with the HSPF model (Pett and Foster, 1985; Brun and Band, 2000; Maximov, 2003) reported that there was a positive correlation between storm runoff volume and the amount of impervious cover. In this study, the simulated land-use change was from agricultural land to forest and wetland, respectively, which might help to explain why the changes in flow volume were not as significant. Third, local site characteristics, such as geographic location, slope, and soils, might also contribute to different flow reductions, as indicated in some BMP studies (Baker and Johnson, 1983; USEPA, 1993; Gitau et al., 2005).

Table 6. Mean Annual and Daily Flow and Nutrients for the Base Case and Various Riparian Land-Use Scenarios (Flow: m^3/s; NO_2+NO_3 and TP: mg/L)

Type	Scenario	Mean annual		Mean daily		Percent Change**
		value	change*	value	change*	
Flow	Base case	1479.73	------	4.05	------	------
	60 m forest buffers	1475.94	-3.79	4.04	-0.01	-0.256%
	60 m wetland buffers	1475.92	-3.81	4.04	-0.01	-0.258%
	90 m forest buffers	1475.87	-3.86	4.04	-0.01	-0.261%
	90 m wetland buffers	1475.87	-3.86	4.04	-0.01	-0.261%
	120 m forest buffers	1475.58	-4.15	4.04	-0.01	-0.281%
	120 m wetland buffers	1475.56	-4.17	4.04	-0.01	-0.282%
NO_2+NO_3	Base case	1591.43	------	4.36	------	------
	60 m forest buffers	1545.40	-46.03	4.23	-0.13	-2.89%
	60 m wetland buffers	1545.60	-45.83	4.23	-0.13	-2.88%
	90 m forest buffers	1519.50	-71.93	4.16	-0.20	-4.52%
	90 m wetland buffers	1519.77	-71.66	4.16	-0.20	-4.50%
	120 m forest buffers	1493.80	-97.63	4.09	-0.27	-6.13%
	120 m wetland buffers	1494.11	-97.31	4.09	-0.27	-6.11%
TP	Base case	37.37	------	0.1023	------	------
	60m forest buffers	36.17	-1.20	0.099	-0.0032	-3.21%
	60m wetland buffers	36.00	-1.37	0.0985	-0.0037	-3.67%
	90m forest buffers	35.50	-1.87	0.0972	-0.0051	-5.01%
	90m wetland buffers	35.24	-2.13	0.0965	-0.0058	-5.70%
	120m forest buffers	34.82	-2.55	0.0953	-0.007	-6.84%
	120m wetland buffers	34.46	-2.91	0.0943	-0.008	-7.78%

*Calculated from current scenario subtracted by base case.
*Percent change = (current scenario - base case) / base case.

3.3. Simulation of the NO_2 + NO_3 and TP Effects of Riparian Land-Use Changes

Compared with the impacts on flow, riparian land use had more effects on the reduction of NO_2 + NO_3 concentration. The 60 m, 90 m, and 120 m riparian forest and wetland buffers reduced the mean annual NO_2 + NO_3 concentration by 2.9 (46 mg/L), 4.5 (72 mg/L), and 6.1% (97 mg/L), respectively (Table 6). Although the impacts from forest buffers and wetland buffers were almost the same, Wilcoxon signed rank tests revealed significant differences for all the 15 paired comparisons (d.f. = 2 556; all $p \leq 0.0001$).

The lowest daily reduction of NO_2 + NO_3 was from 60 m buffers (0.13 mg/L, Table 6), and the highest was from 120 m forest/wetland buffers (0.27 mg/L). According to the USEPA (1986), concentrations of nitrate nitrogen greater than 10 mg/L are considered unsafe for human consumption. The simulated results in this study were all within the safety limit for drinking water supply. Nonetheless, according to Sharma et al. (2008), nitrogen concentration of above 0.5 mg/L is reported to acelerate eutrophication in some water systems. Hence, any reduction in nitrogen will help in ameliorating eutrophication. In this study, it seems that the 120 m buffers can reduce the average daily concentration of NO_2 + NO_3 by 0.27 mg/L, which certainly can help in mitigating the eutrophication problems in the area.

The impacts of riparian land use on TP (Table 6) followed the general trend on flow and nitrogen and had similar magnitudes as those on nitrogen. The 60 m, 90 m, and 120 m riparian forest and wetland buffers decreased the mean annual TP loads by 3.2 to 3.7 % (1.2 to1.4 mg/L), 5 to 5.7 % (1.9 to 2.1 mg/L), and 6.8 to 7.8% (2.6 to 2.9 mg/L), respectively. However, the reductions from forest buffers were slightly lower than those from wetland buffers. Nonparametric tests showed that significant differences were present for all 15 paired comparisons (Wilcoxon signed rank test, d.f. = 2 556; all $p <$ 0.0001).

The decreases of mean daily TP concentrations ranged from 0.003 (60 m forest, Table 6) to 0.008 mg/L (60 m wetland). The base case had a mean daily TP load of 0.102 mg/L, but all of the six scenarios had a mean daily TP loads below 0.1 mg/L. Since the USEPA's suggested drinking water daily limit for TP is 0.1 mg/L (USEPA, 1986), it implies that the buffer strip can help to reduce the mean daily TP loads to the acceptable level.

The HSPF modeling research by Maximov (2003) showed that with the expansion of urban areas, annual concentrations of phosphorus and NO_2 + NO_3 increased by over 36% and 15%, respectively. On the other hand, changing riparian land-use types from agriculture to forest or wetland could reduce the potential of eutrophication and improve water quality. Nevertheless, the effectiveness of nitrogen and phosphorus removal from riparian buffer zones varied widely in the literature. For N, it ranged from 45 (12 m grass buffer, Thompson et al., 1978), 48 (forest buffer, Snyder et al., 1998), to 90% (4.57 m grass buffer, Barfield et al., 1998). For TP, it varied from 1.5 (Perry et al., 1999) to 80 (Hamlett and Epp, 1994) and to 93% (Lee et al., 2000). Although in a smaller magnitude, the results from this study concurred with these findings.

The lower nutrient removal in this study might be attributed to the fact that the headwater of the LMR basin is a well-protected and relatively 'pristine' basin with little or limited anthropogenic impacts. Thus, the impacts of land-use change in the riparian buffers were not as substantial as in other studies. It is also noticed that compared with field studies, model simulation research (Perry et al., 1999; Chen, 2003) generally shows lower reduction percentages for N and P. Often, site specificity is part of the reasons for such great variability (USEPA, 1993; Gitau et al., 2005; Mayer et al., 2006). Moreover, effecttiveness of N and P removal in forested riparian zones can vary widely due to characteristics unrelated to width. For instance, extreme nitrogen loading (Lowrance et al., 1997) and increased hydraulic conductivity of the soil (Pinay and Decamps, 1988; Sabater et al., 2003) may decrease the effecttiveness of forested riparian buffer zones.

Nonetheless, when compared to the flow simulation, this study showed a higher N and P reduction from riparian buffers. If the study area is extended to the entire LMR watershed, it is likely that with increasing riparian forest or wetland area in the buffer zones, more N and P reduction will be achieved.

3.4. Evaluation of BASINS/HSPF

This research has demonstrated that the GIS-based BASINS/HSPF is a reliable and comprehensive water quality and quantity assessment tool. Although HSPF may not be very effective in simulating flow changes in a small watershed, it can still characterize the flow and water quality conditions for the study area and is capable of predicting hydrologic and water quality responses of riparian land-use changes. Besides, BASINS/HSPF has many important advantages, such as ease of use, multi-purpose assessment, model accuracy, and model flexibility. In this research, the five most sensitive hydrologic parameters were INFILT (index to infiltration capacity), INTFW (interflow flow parameter), LZSN (lower zone soil moisture storage), LZETP (lower zone evapotranspiration parameter), and DEEPFR (fraction of groundwater inflow to deeper recharge). These parameters were similar to those identified by Laroche et al. (1996), Carrubba (2000), Engelmann et al. (2002), El-Kaddah and Carey (2004), and Tong and Liu (2006). For the water quality parameters, both Mon-IFLW-CONC and Mon-GRND-CONC had significant impacts on the modeling results. For N simulation, SQOLIM (maximum storage of QUALOF, quality constituents associated with overland flow) and WSQOP (the rate of surface runoff that will remove 90% of stored QUALOF) were moderately sensitive, while for P simulation, POTFS (scour potency factor) was moderately sensitive. However, in modeling the total nitrogen in the Cahaba River, Alabama, El-Kaddah and Carey (2004) found that SQOLIM was moderately to highly sensitive, and WSQOP was moderately sensitive. In the TP modeling study of the Lower Great Miami River basin, Tong and Liu (2006) found that the four most sensitive parameters were SQO (storage of available quality constituent on the surface), SQOLIM, IOQC (concentration of the constituent in interflow outflow), and AOQC (concentration of the constituent in active groundwater outflow). This shows that water quality calibration for HSPF is not only highly dependent on site specific constituents and processes, but also dependent on the watershed characteristics.

4. Conclusions

As an extension of the earlier large watershed studies (Tong, 1990; Liu et al., 2000; Tong and Chen, 2002; Tong et al., 2007), this research attempted to quantify the hydrologic and water quality impacts of riparian forest and wetland buffers in a small headwater subwatershed, the Upper LMR basin. This research is one of the first HSPF-based studies of the hydrologic and water quality impacts of riparian land-use changes under a subwatershed scale. It revealed a few interesting findings. First, riparian land-use change to forests and wetlands even in the headwater subwatershed may be useful in ameliorating the nitrogen and phosphorus concentrations in the receiving water bodies. Second, HSPF is capable of modeling both the water quantity and water quality in this geographical scale. These results might be helpful to our understanding of the plausible and complex relationships between riparian land-use changes and surface water hydrology and water quality. The results could also be used to improve riparian buffer zone restoration practices. As emphasized by Leeds-Harrison et al. (1999), the likely effects of buffer strip on the hydrology should be assessed on a case-by-case basis. This simulation research provides a quantitative, yet expedient and cost-effective, way to evaluate both the hydrologic and water quality consequences of riparian land-use changes, providing more information for developing management solutions.

For future work, buffers for all streams including the higher order ones (44%) in the Upper LMR basin could be created, rather than for just the first and second order rivers (56%). A larger watershed encompassing more study sites downstream of the river with higher nutrient contamination levels would be useful in providing a better picture of the potential effects of land-use changes in riparian buffer zones. Since this headwater study only confined to a small watershed with relatively less contamination, it might not reveal all the possible consequences on water quality. Further research with more subwatersheds and in other geographic areas would help to explore the overall impacts on flow. Other water quality parameters, such as sediment, fecal coliform, and atrazine, could also be examined. More advanced GIS programs could be explored. ArcGIS VBA programs would help to delineate riparian buffers with variable widths based on local topography, climate, and soil characteristics. Besides, this research could be improved using higher resolution DEMs (for example those based on LIDAR-Light Detection And Ranging) and land-use imageries within the simulation period for narrower riparian buffers (less than 60 m). As demonstrated by Baker et al. (2006), riparian metrics (flow-path metrics) calculated based on DEM would help quantify potential nutrient interception by riparian buffers. Groundwater contamination and contaminant transport based on spatial interpolation (Menezes and Inyang, 2009) should also be examined to study the interacttions between surface water and groundwater. It would also be desirable to ascertain HSPF-based modeling results from riparian land-use changes in a small watershed against field studies (Dillaha et al., 1989; Larson et al., 1997). This comparison would provide a better understanding of the interplay of the various factors in the riparian buffers and their combined hydrologic impacts.

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