Modeling the Average Annual Nutrient Losses of Two Watersheds in Indiana Using GLEAMS-NAPRA

R. K. (Mitchell) Adeuya, K. J. Lim, B. A. Engel, M. A. Thomas

Abstract. *From 1973 to 1984 the Black Creek Project assessed the contributions of agricultural production to the agrochemical loading levels to drainage water. GLEAMS-NAPRA calibration and validation were conducted using measured water quantity and quality data from 1975 to 1977 from two watersheds within the Black Creek watershed, the Driesbach and Smith-Fry. The model was calibrated and validated for monthly runoff, nitrate loading, sediment loss, sediment phosphorus, and total phosphorus. Modeling the land use as it existed at the time of the original Black Creek Project using GLEAMS-NAPRA resulted in model predictions that were similar to observed monthly results. In the Driesbach watershed, the Nash-Sutcliffe model efficiencies for monthly runoff, sediment, nitrate, sediment phosphorus, and total phosphorus were 0.89, 0.78, 0.69, 0.57, and 0.70, respectively. Additionally, the R^2 values for monthly runoff, sediment, nitrate, sediment phosphorus, and total phosphorus were 0.90, 0.86, 0.81, 0.79, and 0.75, respectively, for the same watershed. Since the mid-1970s, the land use within the watershed has changed, with an increase in urban and farmstead areas and changes in cropping systems. When applying the model with 2003 land use data, there was a predicted average annual decrease in nitrate loss for the Smith-Fry and Driesbach watersheds by 6 kg/ha and 4 kg/ha, respectively. Little impact was predicted for sediment phosphorus and total phosphorus loss, with differences in average annual loss of 0.05 and 0.08 kg/ha for Smith-Fry and 0.08 and 0.26 kg/ha for Driesbach, respectively. The results of this study indicate that GLEAMS-NAPRA has the ability to predict monthly runoff, nitrate, sediment, sediment phosphorus, and total phosphorus losses on a small watershed scale. The results also indicate the possibility of using GLEAMS-NAPRA to estimate losses in other watersheds with similar soil, land use, and drainage characteristics.*

Keywords. *Drainage efficiency, Land use change, Nonpoint-source pollution, Sediment delivery ratio, Total maximum daily load, Watershed management, Watershed modeling.*

The movement of nitrate is intimately associated with the movement of water. Nitrate, because of its high solubility, is almost everywhere transported in solution (Burt et al., 1993). Of the various forms of nitrogen present in the soil, or added as fertilizer, only nitrate is leached in appreciable amounts by water passing through the soil profile (Wild, 1988). Nitrate leaching and runoff from agricultural land is affected by factors such as fertilization rates. According to Keeney (1986), the greatest problems arise when there is heavy fertilization in intensive row-cropping practices in rain-fed grain production systems such as corn. Bergstorm and Brink (1986) conducted a ten-year study on an arable clay soil in Sweden. They determined that leaching was moderate up to a rate of application of 100 kg N/ha, but increased rapidly, reaching a loss of 91 kg NO_3-N /ha, for the highest application rate of 200 kg N/ha. Barraclough et al. (1983) found that nitrate-nitrogen leached from cut grassland plots, as an equivalent percentage of nitrogen applied, increased from 1.5% to 5.4% to 16.7%, as the fertilizer application increased from 250 to 500 to 900 kg/ha.

Fertilizer-based nitrogen is not the only source of nitrates. In most soils, well in excess of 90% of the nitrogen is present in organic forms. This organic nitrogen is made up of various compounds derived from biological materials such as crop residue and from the humification process (Stevenson, 1982). Mineralization of the organic forms of nitrogen produces nitrates. The mineralization process is driven by the operations of the carbon cycle; as such, this cycle must be taken into account for any analysis of nitrogen processes with the soil and ultimately nitrates.

The application of commercial fertilizer and animal manure on agricultural lands is perhaps the prime source of phosphorus (P). It has been noted that subsurface drainage is potentially the major probable cause for phosphorus losses within a tile-drained watershed (Faruk et al., 2002). Phosphorus, when mobilized, can be in dissolved or particulate form (Shirmohammadi et al., 1998). Although phosphorus can be mobilized as soluble organic P or soluble inorganic P, a greater portion is typically transported as sediment-bound P (Monke et al., 1981).

As indicated in the National Water Quality Inventory and other publications of the U.S. Environmental Protection Agency (USEPA), sediment is the largest nonpoint-source pollutant (USEPA, 2004b). In many watersheds across the U.S., sediment also serves as a contaminant reservoir (USEPA, 1998, 2004a). Additionally, agriculture was identi-

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fied as the leading contributor to water quality impairment in the U.S. due to sediments and nutrients leaving agricultural fields (USEPA, 2004b).

Apart from the application of commercial fertilizers, manure and the processes of nature along with drainage systems play an important role. Artificial drainage systems can have significant impacts on water quality because they behave like shallow direct conduits to surface water (Dinnes et al., 2002). Tile drainage systems are common in the Midwest, covering more than 30% of the crop land (Zucker and Brown, 1998). Drainage is essential to plant growth; however, tile drainage systems expedite the transportation of nitrates to streams and rivers (Davis et al., 2000; Soenksen, 1996; Baker, 1994; Logan et al., 1994). Nutrient loss from an agricultural field also occurs by surface flow or runoff. Nitrate is transported in solution, a product of its high solubility (Burt et al., 1993). Nitrate loss is highly dependent on the volume of runoff, a factor that is affected by rainfall intensity and soil moisture at the time of a rainfall event (Shuman, 2002). The dynamics of drainage systems coupled with fertilizer applications, precipitation, and the processes of the nitrogen cycle make nitrate evaluation a qualified candidate for water quality modeling.

The complexities involved with the interactions of the nitrogen cycle and the carbon cycle, combined with the complexities of nature and with human interventions, suggest modeling as an efficient method for evaluation of situations where these complexities are already considered. Field research is often used to evaluate and acquire knowledge of these processes and their interactions. However, field research can be very costly and time consuming (Davis et al., 2000). Computer simulation models provide efficient and effective tools for analyzing water quality problems (Tim, 1995).

The main objective of this study was to calibrate and test the ability of the GLEAMS model to simulate nitrate losses in surface runoff water from two small agricultural watersheds in northeast Indiana. The second objective was to compare the potential fate of nitrate for land use and cropping systems representing 1975-1977 and 2001-2003 conditions in the two watersheds. The third objective was to determine the average annual sediment, nitrate, and phosphorus losses from the two watersheds, based on model simulations.

GLEAMS-NAPRA Modeling System

Scientists of the USDA-ARS developed the Chemicals, Runoff, and Erosion from Agricultural Management Systems (CREAMS) model (Knisel, 1980), which was later modified to develop Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) model (Knisel, 1993). The GLEAMS model has four components: hydrology, erosion, pesticide, and nutrient. Of the four, the pesticide and nutrient components are optional (Knisel et al., 1992). The GLEAMS model is a mathematical model developed for field-sized areas to evaluate the effects of agricultural management systems on the movement of agricultural chemicals within and through the plant root zone (Leonard et al., 1987). GLEAMS is a one-dimensional, deterministic, and physically based model that simulates percolation, runoff, nutrient and pesticide runoff and leaching, and erosion and sedimentation on a daily time step. It is a complex model that requires many parameters, which can be organized into input files for hydrology, rainfall, nutrients, and pesticides (De Paz and Ramos, 2002). The hydrology file includes soil parameters that can be obtained from the State Soil Geographic Database (STATSGO) and Soil Survey Geographic Database (SSURGO) soil data (http://soils.usda.gov/).

The GLEAMS model is classified as a field-scale model, but it has been used for various applications. Past research supports the ability of GLEAMS to predict the potential level of nitrates in drainage water from agricultural production areas. Leonard et al. (1987) classified GLEAMS as an effective tool to study the critical areas of a watershed. Other research supported GLEAMS as a model that is capable of predicting both water and pesticide leaching in lysimeters (Shirmohammadi and Knisel, 1994). Shirmohammadi et al. (1998) reported that the GLEAMS model is capable of producing reasonable estimates of annual and long-term averages of nitrate and dissolved P to drain tiles. Bakhsh et al. (2000) reported that the GLEAMS model adequately predicts subsurface drain flow and that four-year average nitrate concentrations were in close agreement with measured data.

The National Agricultural Pesticide Risk Analysis (NAPRA) approach that builds on GLEAMS was developed by the USDA-NRCS and the University of Massachusetts to evaluate the complex environmental risks of pesticide use (Bagdon et al., 1994). A WWW-based approach was later developed to estimate site-specific effects of land use and management on water quality, with pesticides being the main concern (Engel et al., 1998). The nutrient component of GLEAMS was added to the NAPRA WWW system to simulate the effects of agricultural management on nutrient water quality (Lim, 1998, 2001; Lim and Engel, 2003). The NAPRA WWW system provides an easy-to-use WWW interface and uses spatial and relational databases to simplify the process of preparing model files (Lim and Engel, 2003). This allows the user to access and run the GLEAMS model from various computer locations.

Nitrate movement in soil and water has been evaluated with the aid of the GLEAMS water quality model in past research. Bakhsh et al. (2000) used the GLEAMS model to compare measured versus simulated effects of swine manure application with urea-ammonium-nitrate on subsurface drain water quality from beneath long-term corn and soybean plots. They found that the model adequately predicted subsurface drain flow and that the four-year average for nitrate was in close agreement with measured data. Bakhsh and Kanwar (2001) calibrated and validated GLEAMS to simulate tillage effects on nitrate-nitrogen and herbicide losses with tile drainage systems beneath continuous corn. Their results showed that simulated nitrate and atrazine losses with tile water were in close agreement with measured data, with a difference of less than 10%. Minkara et al. (1995) conducted a study to investigate the impact of poultry and commercial fertilizer applications on nitrate concentrations below pine seedlings. They found that the trends in the simulated results from the GLEAMS model reflected those measured for surface water and soil nitrate concentrations.

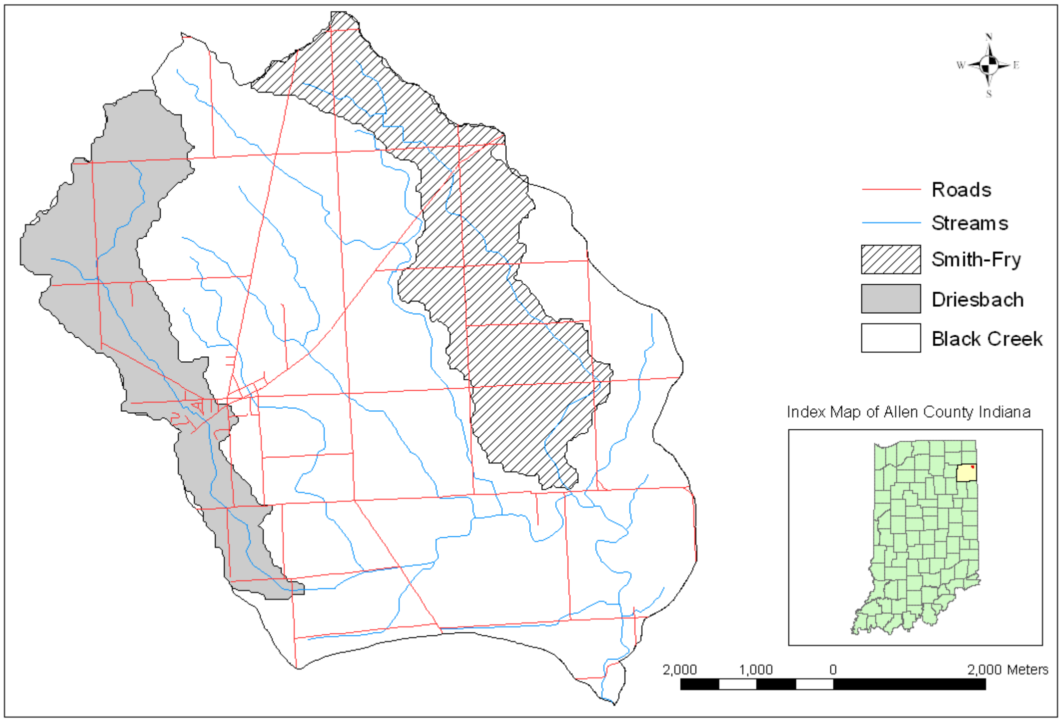

Figure 1. Smith-Fry and Driesbach watersheds.

Methods

Study Area

The Black Creek watershed is approximately 5,000 ha in size and is located in Allen County, Indiana. The watershed is about 24 km northeast of Fort Wayne, Indiana, and is a tributary of the Maumee River, which flows from Fort Wayne to Lake Erie at Toledo, Ohio. The Black Creek watershed was the location of a major research project that was designed to quantify the impacts of agricultural practices on the environment (Lake and Morrison, 1977). The project evolved from concerns about the pollution of Lake Erie and was intended to answer questions on whether agricultural production was a significant contributing factor. The western basin of the lake was considered an environmental disaster in 1972 when the project began. Periodic algal blooms in Lake Erie were believed to be caused by phosphorus carried into the lake (Lake and Morrison, 1983). The study lasted from 1973 to 1984 and employed the use of water quality monitoring stations located at the outlet of the Smith-Fry and Driesbach watersheds. The samplers provided continuous flow data and permitted the calculation of loadings on a storm and time period basis (Nelson et al., 1981). The data from these stations include flow, rainfall, and nutrient concentration levels from about four years of the study (1974-1978).

The Smith-Fry and Driesbach watersheds are 942 ha (2327.7 acres) and 714 ha (1764.3 acres) in size, respectively (fig. 1). The soil type (table 1) in both watersheds consists of nearly level lake plain and beach ridge soils and gently sloping glacial till soils (Nelson et al., 1981; Mitchell, 2004). The soils in these two watersheds are of the Hoytville-Nappanee association that are deep, somewhat poorly drained, nearly level soils of a medium to fine texture (USDA-SCS, 1969). The watersheds are tile drained and predominantly row cropped. Land use types are shown in table 2. The land owners in the Driesbach and Smith-Fry watersheds can be placed in two basic groups, non-Amish and Amish farmers. Amish farmers occupy over 80% and 5% of the land in the Driesbach and Smith-Fry watersheds, respectively.

Table 1. Driesbach and Smith-Fry soil types by percent of watershed area.

Soil Type	Driesbach (%)	Smith-Fry (%)
Fine sandy loam	4.0	2.0
Loam	9.6	29.0
Silt loam	57.0	33.0
Silty clay	2.8	9.0
Silty clay loam	26.6	27.0

Data Sources and Management Practices

ArcGIS 8.3 was used to develop 1975-1977 land use maps from aerial photographs, maps, and 35 mm color slides. The images were rectified to the watershed boundaries and digitized using the editor tool and the auto complete option. The 2001-2003 crop system was developed using 2003 aerial photographs from the USDA-NRCS and the 2002 NASS land use data (USDA-NRCS, 2003). The SSURGO data and the land use, drainage, and ownership data were intersected to

Table 2. Smith-Fry and Driesbach land use classes by percent of watershed area.

	Driesbach (%)			Smith-Fry (%)		
Land Use	1977	1978	1979	1977	1978	1979
Corn	23.6	26.6	32.4	29.6	27.3	34.4
Farmstead	8.6	8.6	8.6	6.8	6.8	2.6
Pasture/grass	38.7	35.4	28.3	6.2	4.9	8.0
Small grain	13.3	8.1	16.4	22.3	17.4	15.8
Soybean	6.5	12.0	5.0	28.8	37.2	31.2
Urban/residential	4.1	4.1	4.1	0.0	0.0	0.0
Woodland	5.2	5.2	5.2	6.4	6.4	8.0

identify the unique combinations of soil, land use, drainage, and ownership in the Smith-Fry and Driesbach watersheds.

The nutrient and pesticide management practices were varied within the study according to the owner, whether Amish or non-Amish. The nutrient and pesticide management practices for areas occupied by the Amish were taken from historical data (Christensen and Wilson, 1975). For the non-Amish owned areas, the Indiana state averages were used (USDA-SCS, 1975-1978).

The recorded rainfall data for January 1974 to June 1977 were located in the historical Black Creek project data (Lake and Morrison, 1977). The long-term rainfall data file was created using NAPRA WWW, and the precipitation file was adjusted to include the recorded data for January 1974 through June 1977. The temperature, radiation, and other climatic data for the same time period were obtained from the Allen County weather station data stored in the NAPRA WWW database.

Model Calibration

Hydrology

The historical flow data were divided in two parts: the first half of the data (1975-1976) was used for calibration of hydrology, and the second half (1977-1978) was used for validation. The rainfall data used for the Smith-Fry watershed were applied to the Driesbach watershed during the validation process. The rainfall for the 1975-1976 (1968 mm) and 1977-1978 (1686 mm) periods had very similar trends (Mitchell, 2004). GLEAMS was calibrated using observed data from the automatic water quality samplers at the outlet of Smith-Fry and Driesbach. Calibrating the hydrology was the first step in the total calibration process. Some parameters (table 3), including curve number, sediment delivery ratio, and drainage efficiency, were calibrated using software written for this purpose, while the remaining parameters were calibrated through manual techniques (Mitchell, 2004). A summary of the input parameters used for calibration is shown in table 3. The movement of water through the system greatly affects nutrient transport, especially for nitrate. The original data recorded as stream flow had to be separated into base flow and runoff using the digital filter method available at the Web-based Hydrograph Analysis Tool (WHAT) (Lim et al., 2004; http://pasture.ecn.purdue.edu/~what).

A tile drainage efficiency factor was used to account for the movement of water leached below the root zone through the tile drainage system. An efficiency factor range of 0 to 1.0 was used to find the drainage efficiency that best represented both watersheds. Other studies with tile drainage reported a drainage efficiency of 0.866 (Davis et al., 2000). The research conducted by Davis et al. (2000) modeled field and predicted estimated components of the water balance. Of the water moving below the root zone (tile drainage and deep seepage), tile drainage accounted for 0.866.

The curve numbers for each land use type and hydrological soil group used in the calibration were obtained from the GLEAMS 3.0 user manual (Knisel and Davis, 1999). The curve number was calibrated using the minimum, average, and maximum curve numbers for each land use and soil combination to determine which curve number regime provided results that were a better fit to the observed historical data. The minimum curve number provided results that were the best fit to the observed data. The drainage efficiency and curve number combination that resulted in the minimum difference between simulated and observed data were used for validation. Drainage efficiencies from 0.5 to 1.0 were used in calibration. The drainage efficiency that resulted in simulated data that had the best match to the observed data was selected. A drainage efficiency of 0.75 was chosen, because it resulted in simulated monthly and total runoff for the time period that had minimum difference with observed values for the same period.

The coefficient of determination (R^2) and the coefficient of efficiency (or Nash-Sutcliffe model efficiency - NSE) (Nash and Sutcliffe, 1970) were used to evaluate model predictions for both calibration and validation. The R^2 value indicates the strength of the relationship between the observed and simulated values; whereas the NSE indicates how well the observed versus the simulated value fit the 1:1 line (Santhi et al., 2001). The closer these values are to 1, the more favorable the simulation. If values are close to zero, that indicates a poor relationship between observed and simulated data. The R^2 and NSE values for hydrology suggested by Santhi et al. (2001) for acceptable model performance are 0.7 and 0.6, respectively, while Ramanarayanan et al. (1997) suggests that model prediction is acceptable if NSE is greater than 0.4 and R^2 is greater than 0.5. NSE greater than 0.45 and

Table 3. GLEAMS model inputs selected during calibration.

Parameter	Description	Model or Established Range	Calibrated Value
DE	Drainage efficiency of tile drains	0-1.0	0.75
CN	Curve number	8-93	25-90
NFACT	Manning's N for overland flow profile	0.01-0.40	0.03-0.05
PFACT	Universal Soil Loss Equation (USLE) contouring factor for overland flow profile	0.01-1.0	0.3
CFACT	Soil loss ratio representing the USLE crop factor for overland flow profile	0.1-1.0	0.19-0.75
SDR	Sediment delivery ratio	0.2-0.5	0.3
TN	Total nitrogen percent in soil horizon	0-10	0.2
CNIT	Nitrate-nitrogen concentration in soil horizon (μg/g)	0-1000	30
POTMN	Potentially mineralizable nitrogen in soil horizon (kg/ha)	0-1000	262, 50, 35, 25, 20
ORGANW	Organic nitrogen from animal waste in plow horizon (%)	0-100	0.02
TP	Total phosphorus in the soil horizon (%)	0-10	0.002
CLAB	Labile-phosphorus concentration in the soil horizon (μg/g)	0-1000	6-10
ORGPW	Organic phosphorus content from animal waste in the plow horizon (%)	0-10	0.015
Nitrate application rate	Percent increase or decrease in the amount of nutrient applied	0-100	20-50
Phosphorus application rate	Percent increase or decrease in the amount of nutrient applied	0-100	10

R^2 greater than 0.50 were selected as the evaluation criteria for calibration and validation in this study.

Sediment

For sediment calibration, the Universal Soil Loss Equation (USLE) P-factor (conservation practices factor) was adjusted to account for best management practices in the watersheds. A range of P-factors was used, with P-factors ranging from 0 to 1. A P-factor of 0.3 was identified for use through calibration (table 3). The USLE C-factor (cover and management factor) was adjusted with ranges above and below the values suggested by USDA-SCS (1972b) and Walters et al. (1988). The USLE C-factors and P-factor were calibrated for the Smith-Fry watershed and then applied to the Driesbach watershed during the validation process (Mitchell, 2004).

One other parameter that was used in calibration was sediment delivery ratio (SDR). The SDR is used to estimate the amount of erosion from a watershed that reaches the outlet. For the purpose of this research, three methods were used to determine a potential range in values for SDR. The three equations are as follows:

$$SDR = 0.4724A^{-0.125} \text{ (Vanoni, 1975)}$$

$$SDR = 0.3750A^{-0.2382} \text{ (Boyce, 1975)}$$

$$SDR = 0.5656A^{-0.11} \text{ (USDA-SCS, 1972a)}$$

where A is the watershed area (km^2).

These equations helped to establish an SDR range of 0.20 to 0.50 for the watersheds in this study. The model was calibrated using this range, with a sediment delivery ratio of 0.3 providing the best results for calibration.

Nitrate

Based on the hydrology calibration, a drainage efficiency ratio was established for use in nitrate calibration. The drainage efficiency of 0.75 and minimum curve number values, as suggested by Knisel and Davis (1999), were used for nitrate calibration. The parameters that were established during the sediment calibration remained the same for nitrate calibration. Nitrate concentration was calibrated using recorded nutrient data from 1975-1977. The initial soil concentration levels of nutrients were unknown, so default GLEAMS values were used. During the calibration process, changes in soil nutrient concentration parameters did not improve simulated nutrient losses for the watershed. However, varying nutrient application rates resulted in significant improvements in outputs. Nutrient applications were unknown, so a ±50% range was attached to initially estimated nutrient application levels. Nitrogen application rates, in the form of urea, for calibration were 50% more per application for non-Amish and 20% more per application for Amish farmers than the estimated historical application values (Mitchell, 2004). For corn, the non-Amish farmers added 128 kg/ha according to historical estimates, while the Amish farmers added 54 kg/ha. During calibration, the application rates for non-Amish and Amish farmers were 193 and 65 kg/ha, respectively (for details on other crops, see Mitchell, 2004).

Phosphorus

Two forms of phosphorus losses were calibrated, sediment P and total P. Phosphorus loss was calibrated using all the parameters that were set in the hydrology, sediment, and nitrate calibrations. The calibration period for phosphorus was January 1976 to May 1977. There was a ±50% range attached to initial estimates of phosphorus application rates. The calibrated rate of application for phosphorus, in the form of triple superphosphate, was established to be 10% lower per application for both non-Amish and Amish farmers than the historical recorded application rates.

MODEL VALIDATION

Model validation was performed to test the accuracy of the model in predicting values for observed data that were not included in the calibration process. The parameters adjusted during calibration (table 3) were held constant during validation. The model predictions were compared to the observed data. Predicted and observed data were analyzed using the same statistical procedure as with calibration to determine how well the predicted values compared to the observed values for the same time periods. The calibration periods for flow and nutrients differ because of limited observed data for nutrients. Flow data were available for 1975 to 1978, while only two and a half years (January 1975 to May 1977) of reliable observed nutrient and sediment data were available. Flow was validated using data from both Smith-Fry and Driesbach, while sediment, nitrate, and phosphorus were validated using data from the Driesbach watershed only. The assumption was made that since Smith-Fry and Driesbach predictions have similar responses in validation of hydrology, then the responses for sediment and nutrients would be similar.

LAND USE COMPARISON

The GLEAMS model was run for 2003 land use data using the calibrated model parameters for the historical time period. The crop system associated with 2003 was developed using NASS 2002 land use data and 2003 aerial photographs from the USDA-NRCS. Crop rotation for a three-year period was established and used for simulation. The aim was to assess the influence of the change in land use on the loading levels of nitrate, sediment P, and total P. The simulated results using the 2003 land use and cropping system data were compared to the historical simulated results for a cropping period of three years (1975-1977). The years 1975-1977 were chosen as the years for comparison because the parameters were calibrated during that time period. The management practices and fertilizer application rates were assumed to be the same for the 1975-1977 and the 2001-2003 periods. Therefore, the only difference between the 1975-1977 and the 2001-2003 simulations was land use; all other factors and model parameters remained constant.

RESULTS AND DISCUSSION

MODEL CALIBRATION AND VALIDATION

GLEAMS-NAPRA calibration and validation results were satisfactory, as shown by the Nash-Sutcliffe model efficiency (NSE) and correlation coefficient (R^2) values obtained (tables 4 and 5). An NSE value greater than 0.45 was computed for all variables during the calibration and validation period.

Runoff

Runoff calibration results for Smith-Fry resulted in NSE and R^2 values of 0.62 and 0.70, respectively, with a difference

Table 4. Monthly calibration and validation results for Smith-Fry watershed.

Variable	Obs. Mean	Pred. Mean	NSE	R^2	Time Period
Calibration					
Runoff (mm)	16	17	0.62	0.70	Jan. 1975 - Dec. 1976
Sediment (t/ha)	0.042	0.042	0.71	0.73	Jan. 1976 - May 1977
Nitrate (kg/ha)	1.05	1.06	0.67	0.67	Jan. 1975 - May 1977
Sediment P (kg/ha)	0.06	0.05	0.49	0.77	Jan. 1976 - May 1977
Total P (kg/ha)	0.10	0.073	0.54	0.58	Jan. 1976 - May 1977
Validation					
Runoff (mm)	15	16	0.88	0.89	Jan. 1977 - Nov. 1978

Table 5. Monthly validation results for Driesbach Watershed.

Variable	Obs. Mean	Pred. Mean	NSE	R^2	Time Period
Runoff (mm)	16	17	0.89	0.90	Jan. 1977 - Nov. 1978
Sediment (t/ha)	0.02	0.018	0.78	0.86	Jan. 1976 - May 1977
Nitrate (kg/ha)	0.70	0.50	0.69	0.81	Jan. 1975 - May 1977
Sediment P (kg/ha)	1.20	0.81	0.57	0.79	Jan. 1976 - May 1977
Total P (kg/ha)	0.70	0.60	0.70	0.75	Jan. 1976 - May 1977

in the monthly mean of 1 mm (table 4). The predicted runoff values for Smith-Fry had a similar trend to the observed data, with the model overpredicting runoff in January 1975, July 1975, and February 1976.

Runoff validation for Smith-Fry resulted in NSE and R^2 values of 0.88 and 0.89, respectively, with a difference in the monthly mean of 1 mm (table 4). Although the model over- or underpredicted runoff in some months (fig. 2), the trends in predicted values were similar to the observed values.

The Driesbach watershed was used for further validation of the model. The predicted values had an excellent match to the observed monthly values, resulting in NSE and R^2 values of 0.89 and 0.90, respectively (table 5). The model results showed a slight overprediction in January and February of 1977 and a slight underprediction in March 1978 (fig. 3).

The results of the Driesbach runoff validation indicate that, although the model was calibrated on the Smith-Fry watershed, there were better results when calibrated parameters were applied to the Driesbach watershed (tables 4 and 5). This suggests similarities in the hydrology of Smith-Fry and Driesbach and supports the potential use of GLEAMS-NAPRA to make predictions on watersheds with similar characteristics using the same calibrated values.

Sediment

Sediment calibration for Smith-Fry resulted in NSE and R^2 values of 0.71 and 0.73, respectively, with the same monthly mean of 0.042 t/ha (table 4). The model underpredicted monthly sediment loss in February 1976 while overpredicting losses in February and March of 1977 (Mitchell, 2004). The values between the two peaks of February 1976 and March 1977 (fig. 4) in the data are very small and can be attributed to low runoff volume for that time (Mitchell, 2004). In general, the model simulated sediment loss effectively, with the exception of the underpredictions in peak flow months, which is common when applying

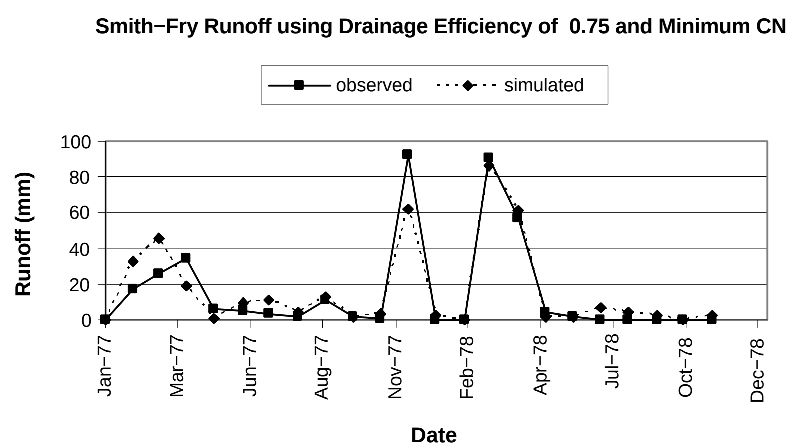

Figure 2. Monthly Smith-Fry watershed validated runoff values for 1977-1978.

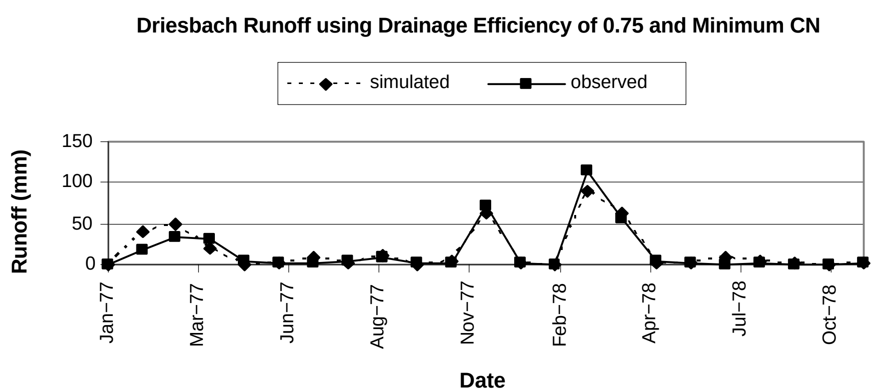

Figure 3. Monthly Driesbach watershed validated runoff for 1977-1978.

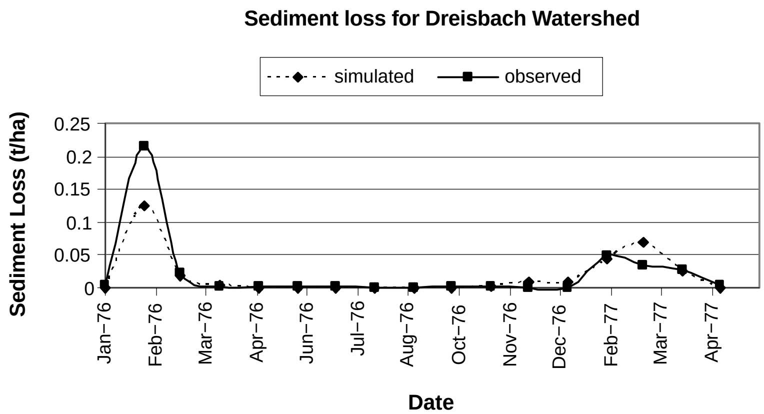

Figure 4. Monthly Driesbach watershed validated sediment loss for 1976-1977.

long-term continuous simulation models during months with high flow volumes (Borah and Bera, 2004).

The validation for sediment estimation was done using the Driesbach watershed and resulted in NSE and R^2 values of 0.78 and 0.86, respectively (table 5). The model underpredicted sediment loss in February 1976 and overpredicted losses in February and March of 1977 (fig. 4).

Sediment calibration and validation results were acceptable. The statistical results for Driesbach could have been even better if a higher sediment delivery ratio was used. The sediment delivery ratio is a function of watershed size, based on the equations applied for calculation. Driesbach is about 200 ha smaller than Smith-Fry. Although these watersheds may have similar characteristics, the amount of sediment reaching the outlet may be more for Driesbach than for Smith-Fry because of a higher sediment delivery ratio, a factor that is typically affected by watershed size. A smaller watershed will potentially have a higher proportion of sediment that is expected to reach the outlet. This may be a limitation to using Driesbach for validation, and it should be considered when making predictions on watersheds with similar characteristics but different sizes. The sediment delivery ratio may also be affected by other factors, including drainage density, slope, land use, and surface roughness. The approach used within this study did not consider the influence of these watershed characteristics on sediment delivery.

Nitrates

Nitrate loss calibration for Smith-Fry resulted in NSE and R^2 values of 0.67, with a difference in the monthly mean of 0.01 kg/ha (table 4). The nitrate calibration data had a similar trend to the observed data, with overpredictions of nitrate loss in May 1975 and February 1976 and underprediction in March 1977 (Mitchell, 2004). The validation data for Driesbach resulted in NSE and R^2 values of 0.69 and 0.81, respectively, with a difference in monthly mean of 0.2 kg/ha (table 5). The Driesbach validation values for nitrate loss had a similar trend to observed data, except for underprediction in January to March 1975 and in February 1977 (fig. 5).

The nitrate predictions by GLEAMS-NAPRA were acceptable, as evidenced by the high statistical correlation and agreement values calculated. The uncertainty associated with nitrogen and phosphorus simulations is a function of the appropriateness of the values selected to describe the application rates of nutrients, the tillage systems, and the initial soil nutrient levels. For instance, the application rates of nutrients as developed from the Black Creek files and the Indiana Agricultural Statistics Service may not accurately capture what individual farmers practiced on their farms. One nutrient application rate was assumed for all non-Amish farmers and another for all Amish farmers. In reality, rates may vary substantially from farm to farm, as could tillage systems and initial soil nutrient levels. Further, the exact dates of nutrient applications were unknown and therefore were assumed. Also of interest are the similarities in the peaks of nitrate loss and runoff (figs. 3 and 5). The similarities indicate that nitrate losses from the watersheds are highly influenced by runoff, as would be expected. The similarities between the nutrient losses for calibration and validation

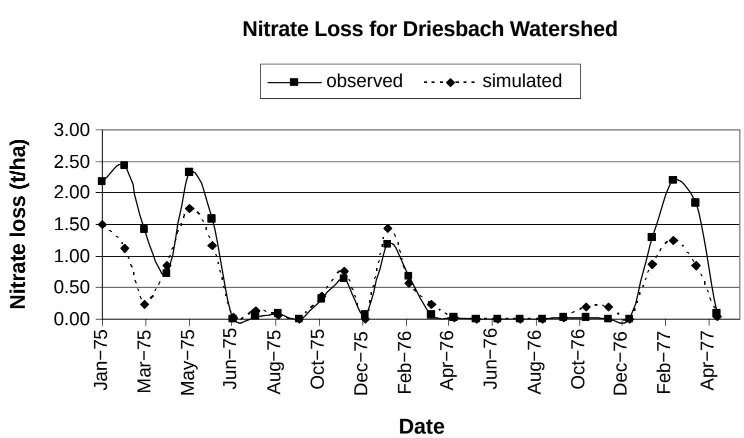

Figure 5. Monthly nitrate in runoff validation results for Driesbach watershed for January 1975 to May 1977.

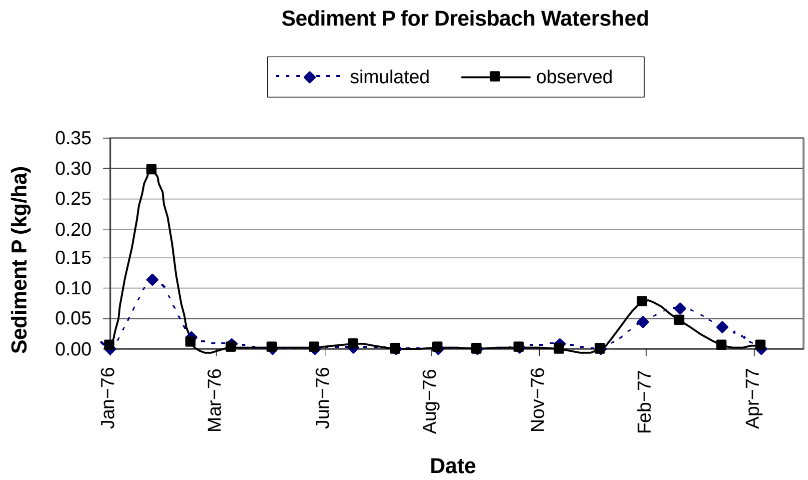

Figure 6. Monthly sediment phosphorus loss validation for Driesbach watershed for January 1976 to May 1977.

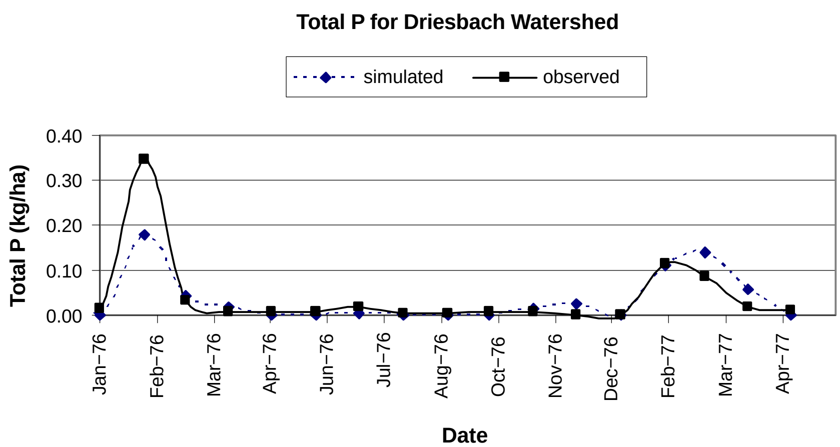

Figure 7. Monthly total phosphorus loss validation for Driesbach watershed for January 1976 to May 1977.

suggest that the nutrient application rates obtained through calibration are reasonable.

Phosphorus

Calibrations for Smith-Fry resulted in NSE and R^2 values of 0.49 and 0.54 for sediment P and 0.77 and 0.58 for total P, respectively (table 4). Both sediment P and total P were underpredicted by the model in February 1976 and overpredicted in February 1977 (Mitchell, 2004). Validation values for Driesbach resulted in NSE and R^2 values of 0.57 and 0.70 for sediment P and 0.79 and 0.75 for total P, respectively (table 5). The model underpredicted both sediment P and total P for the Driesbach watershed in February 1976 and had slight overpredictions in March 1977 (figs. 6 and 7).

All phosphorus simulations resulted in values close to zero in April to December of 1976, a trend that is very similar to observed values for the sediment calibration and validation for this same time period (figs. 4, 6, and 7). These results provide evidence of a strong relationship between sediment movement and phosphorus loss from a watershed.

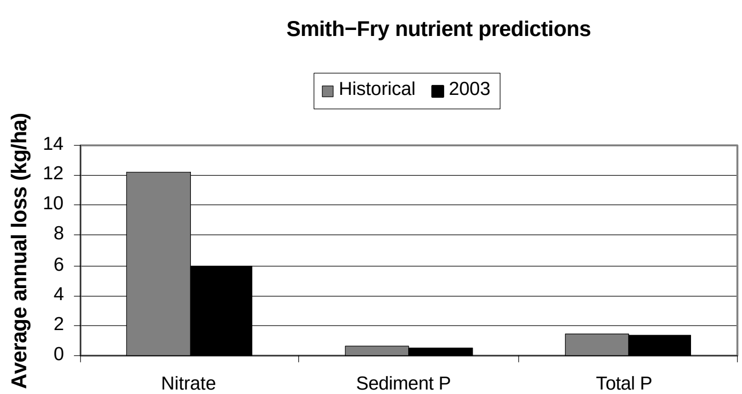

Figure 8. Comparison of annual nutrient predictions for Smith-Fry watershed.

IMPACT OF LAND USE CHANGES

The land use changes within the Smith-Fry watershed include a 4% decrease in row crops, no change in small grain/pasture, a 1% increase in forest, and a 3% increase in urban/roads (Mitchell, 2004). The changes in land use for the Smith-Fry watershed from the 1975 to the 2003 cropping systems resulted in a decrease in the average predicted annual nitrate loss of over 6 kg/ha (51%) and an average annual decrease in sediment P and total P loss of 0.05 and 0.08 kg/ha (7% and 6%), respectively (fig. 8 and table 6). The reduction in the estimated nitrate losses was attributed to the reduction in row crops and the increase in forest and urban/roads that occurred on the watershed.

The land use changes within the Driesbach watershed include a 13% decrease in row crops, a 6% increase in small grain/pasture, a 2% increase in forest, and 5% increase in urban/roads (Mitchell, 2004). The changes in land use for the Driesbach watershed resulted in a decrease in the predicted average annual nitrate loss of over 4 kg/ha (40%) and an increase in predicted sediment P and total P loss of 0.08 and 0.26 kg/ha (26% and 45%), respectively (fig. 9 and table 6). The 13% decrease in row crops combined with the 6% increase in small grain/pasture and the 2% increase in forest is largely responsible for the decrease in nitrate levels. Sediment P and total P seemed to have been increased only slightly by the land use change.

Table 6. Predicted average annual nutrient losses for Smith-Fry and Driesbach watershed for 1975-1977 and 2001-2003.

Nutrient (kg/ha)	Smith-Fry		Driesbach	
	1975-1977	2001-2003	1975-1977	2001-2003
Nitrate	12.18	5.94	10.52	6.29
Sediment P	0.59	0.55	0.31	0.39
Total P	1.42	1.34	0.58	0.84

SUMMARY AND CONCLUSIONS

The 1975-1977 land use data for the Smith-Fry and Driesbach watersheds were developed from aerial photographs and historical land use maps using GIS. A comprehensive data set was developed that included owner (established farming practices), land use, drainage (tiled or non-tiled), and soil type to effectively capture the characteristics of the Smith-Fry and Driesbach watersheds as they existed in 1975-1977. GLEAMS-NAPRA was able to simulate the 1975-1977 water quality data and predicted values that were very similar to observed data, evidenced by NSE values greater than 0.45 and R^2 values greater than 0.5 on a monthly basis for all simulated parameters.

The simulations revealed that watersheds with similar characteristics, in terms of hydrology, cropping systems, drainage, and nutrient application rates, can be used for calibration and validation of the model. Additionally, the results indicated that the use of similar watersheds for validation may be limited by the sediment delivery ratio, a factor typically influenced by watershed size. During calibration of the model, a drainage efficiency of 0.75 was established for the Smith-Fry and Driesbach watersheds, suggesting that approximately 75% of water and NPS pollutants moving below the root zone are intercepted by subsurface drains in these watersheds.

The comparison of simulated nutrient losses for historical versus 2003 cropping system and land use changes showed a decrease in row crop production for the Smith-Fry and Driesbach watersheds of 4% and 13%, respectively; this resulted in a predicted reduction in average annual nitrate loss of 6.0 kg/ha (51%) and 4.0 kg/ha (40%), respectively. The changes in land use also resulted in a predicted increase in average annual sediment P and total P loss for Driesbach of 26% and 45%, respectively. The results for Smith-Fry showed a reduction in the predicted sediment P and total P loss of 7% and 6%, respectively.

The results of the study indicate that the field-scale model GLEAMS can be successfully applied to small watersheds (nearly 1000 ha). Therefore, GLEAMS provides a consistent approach for addressing agricultural NPS pollution issues at scales ranging from field to small watershed. Since there were no nutrient monitoring data from the 1990s or 2000s for the study watersheds, this study points to the opportunity to use water quality models, such as GLEAMS-NAPRA, in examining the impacts of changes in watersheds. Further studies could be done to acquire measured data to confirm the changes in water quality for the two watersheds, as influenced by land use changes and cropping systems. Such studies may further support the use of GLEAMS-NAPRA in estimating agricultural NPS pollutant losses from watersheds as well as fields.

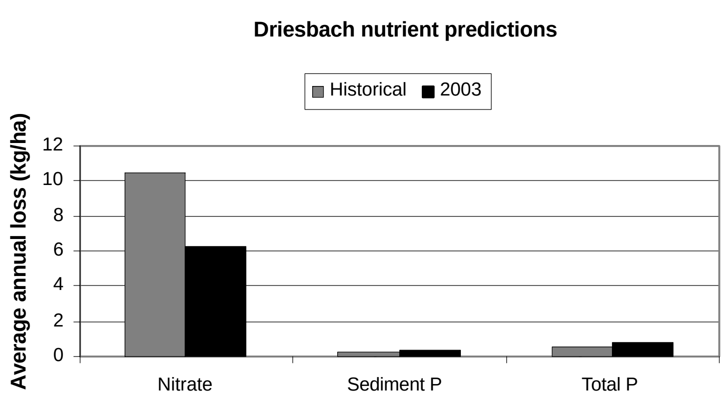

Figure 9. Comparison of annual nutrient predictions for Driesbach watershed.

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