Journal of Spatial Hydrology
Vol.3, No.1 Fall 2003

A SPATIAL TECHNIQUE FOR ESTIMATING STREAMBANK ROSION BASED ON WATERSHED CHARACTERISTICS

*Barry M. Evans, Scott A Sheeder, and David W. Lehning

Penn State Institutes of the Environment
The Pennsylvania State University
University Park, PA 16802

*Corresponding author: bme1@psu.edu

ABSTRACT: A GIS-based technique was developed for estimating streambank erosion rates for more accurately predicting total sediment loads at the watershed scale without the use of detailed field data. This technique relies on the use of data sets that are easily obtained and expressed as GIS data layers. The basis of this technique are statistical relationships between "lateral erosion rates" and watershed characteristics such as curve number, grazing animal density, topographic slope, soil erodibility, and degree of urban development. An algorithm for estimating streambank erosion was incorporated into a GIS-based watershed model. Simulated and observed sediment loads were compared for twenty-eight watersheds in Pennsylvania, and a relatively good model fit was obtained based on a number of statistical measures.

KEY WORDS: GIS, watershed modeling, streambank erosion, sediment transport

INTRODUCTION

As mandated by the Clean Water Act of 1977, states and territories of the U.S. are required to conduct Total Maximum Daily Load (TMDL) assessments for all surface water bodies deemed to be impaired by point and/or non-point source pollutants. Essentially, such an assessment involves the estimation of the maximum amount of a pollutant (or pollutants) that can enter a water body and still meet water quality standards, as well as the allocation of that amount to various pollutant sources (U.S. Environmental Protection Agency, 2000). In Pennsylvania, as in many other areas around the country, various water quality models are being used to support TMDL assessments. In impaired watersheds where non-point pollution sources are the primary issue, the Pennsylvania Department of Environmental Protection (PaDEP) has selected AVGWLF, a GIS-based watershed modeling tool developed by Evans et al. (2002) as its preferred modeling approach. AVGWLF provides an interface between ArcView GIS software and the Generalized Watershed Loading Function (GWLF) model developed by Haith and Shoemaker (1987).

During the initial testing of AVGWLF in selected watersheds in Pennsylvania, it was noted that that the original GWLF model did not include algorithms for estimating sediment generated within a given watershed via streambank erosion. Sediment produced via this process can be quite significant, especially in watersheds that are highly urbanized, have steep slopes, and/or have large grazing animal populations with unimpeded access to streams (Dietrich and Dunne, 1978; Novotny and Olem, 1994; Nelson and Booth, 2002; Trimble, 1994; and Williamson et al., 1992). Unfortunately, the mechanisms controlling the rate of streambank erosion and sediment transport are extremely complex, and therefore difficult to model with accuracy. Many attempts have been made to estimate soil loss from streambank erosion by developing complex models or approaches that require site-specific data on stream bank characteristics and morphometry that require extensive field data collection (e.g., Green et al., 1999; and Sekely et al., 2002). However, the resource-intensive nature of such approaches makes their use infeasible when the assessment of a large number of watersheds is required or when time and funds are extremely limited.

In this study, algorithms for estimating soil losses from streambank erosion within a watershed were incorporated into AVGWLF. These algorithms rely on the use of relatively easy-to-obtain data that can be readily expressed as GIS data layers, and are based on an empirical method commonly used by researchers in hydro-geomorphic studies. This method was subsequently tested and refined in a number of watersheds in Pennsylvania.

METHODOLOGY

Empirical Model for Estimating Streambank Erosion

The empirical model used in estimating soil loss from streambank erosion is based upon the familiar sediment transport function having the form

$$C = aQ^b \tag{1}$$

where Q is stream discharge for some time period, C is either suspended sediment concentration or yield, and a and b are empirically-derived values (VanSickle and Beschta, 1983; Lemke, 1991). In this particular case, a revised version of this equation is used where C represents the "lateral erosion rate". This refers to the total distance that soil is eroded away from both banks along the entire length of a stream during a specified period of time. Using this concept, sediment loads from stream bank erosion are estimated within AVGWLF using a two-step procedure. First, the lateral erosion rate is estimated using the equation

$$LER = aQ^{0.6} \tag{2}$$

where LER is the lateral erosion rate in m/month, a is an empirically-derived "erosion potential" factor, and Q is mean monthly stream flow in m^3/sec. In this case, the value of 0.6 used for factor b is the value recommended by Rutherford (2000) based on a global review of stream bank erosion studies. The resultant LER value is then multiplied by the total length of streams in the watershed (in meters), an estimate of average stream bank height (in meters), and average soil bulk density (in kg/m^3) in order to calculate monthly sediment loads generated by streambank erosion.

Model Parameter Estimation

The value of the empirically-derived "a" constant is related to a wide variety of watershed characteristics such as the amount of infiltration, runoff, inherent soil erodibility, amount of rainfall, and other watershed-related factors (Prosser et al., 2001; and Rutherford, 2000). Consequently, this constant can differ greatly from watershed to watershed. Based upon a review of global studies, Rutherford (2000) has suggested a median value of approximately 0.016 for calculating annual streambank-eroded loads, with values ranging from about 1×10^{-7} to 1×10^{-1}. Similarly, Dietrich et al. (1999) calculated this constant to have a value of approximately 0.008 for estimating annual streambank sediment loads for selected rivers in Australia.

For the present study, monthly lateral erosion rates were estimated by comparing observed sediment loads for twenty-eight different watersheds (see Figure 1) against GWLF-simulated loads for the same time period. Although drawn from a very limited pool of watersheds having historical sediment load data, these watersheds do reflect a wide variety of landscape conditions found within Pennsylvania (Table 1).

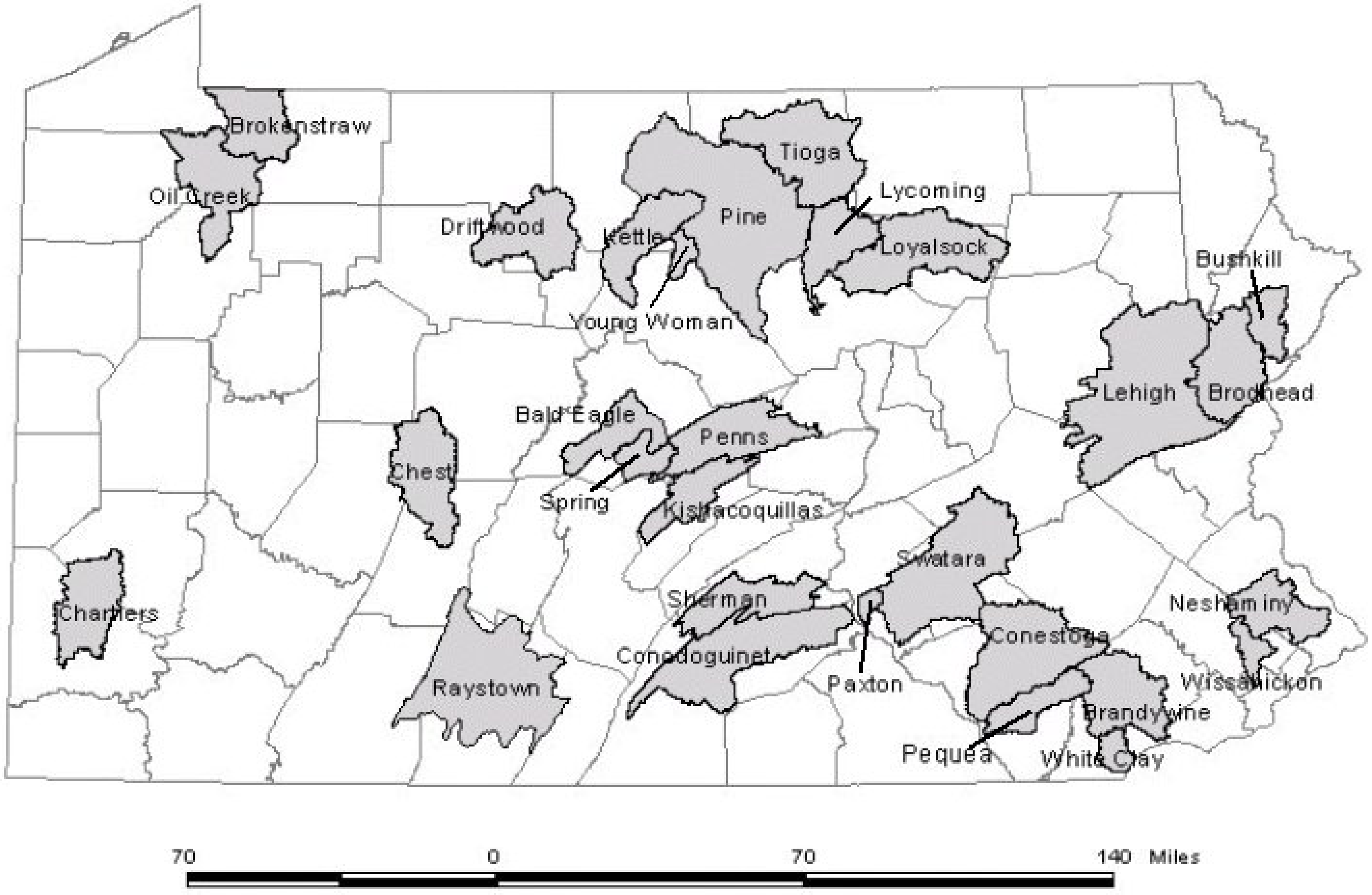


Figure 1. Location of watersheds used for streambank erosion calculation.

Table 1. Landscape characteristics by watershed.

Watershed Name	Area (hectares)	Percent Wooded	Percent Disturbed[1]	Percent Agriculture	Percent Developed	Mean CN	Mean K Factor	Mean Slope (%)	Animal Density[2]
Bald Eagle Creek	68,635	60.3	0.3	35.6	3.8	75.7	0.247	9.4	0.073
Brandywine Creek	74,908	43.9	2.3	46.7	7.1	66.9	0.314	3.1	0.127
Brodhead Creek	77,864	87.4	0.0	8.2	4.4	71.6	0.242	6.0	0.005
Brokenstraw Creek	60,446	75.8	0.5	22.9	0.8	70.4	0.310	6.2	0.050
Bushkill Creek	30,426	98.5	0.3	0.5	0.7	71.4	0.230	4.1	0.000
Chartiers Creek	71,181	48.9	1.0	32.6	17.5	76.6	0.351	5.7	0.055
Chest Creek	81,620	77.6	2.9	18.5	1.0	75.0	0.295	6.9	0.027
Conestoga River	122,969	25.0	1.0	64.3	9.7	73.1	0.312	3.0	0.773
Conodoguinet Creek	131,270	32.8	0.7	61.3	5.2	76.2	0.260	4.4	0.164
Driftwood Branch	77,094	96.5	0.5	2.6	0.4	67.9	0.242	14.1	0.003
Kettle Creek	63,810	95.9	0.3	3.2	0.2	70.2	0.227	15.3	0.003
Kishacoquillas Creek	48,340	61.7	0.3	35.9	2.1	75.7	0.236	11.9	0.207
Lehigh River	228,447	84.3	1.7	11.5	2.5	74.5	0.231	8.9	0.020
Loyalsock Creek	113,146	88.6	1.0	10.1	0.3	73.3	0.243	10.2	0.031
Lycoming Creek	55,805	85.6	0.4	13.4	0.6	73.8	0.236	14.0	0.053
Neshaminy Creek	53,940	37.6	1.1	41.1	20.2	77.4	0.372	1.8	0.018
Oil Creek	84,536	76.9	0.4	21.8	1.0	74.2	0.320	4.8	0.050
Paxton Creek	7,097	32.4	0.2	28.6	38.8	80.5	0.237	3.6	0.013
Penns Creek	80,353	70.4	0.4	28.8	0.4	74.3	0.222	10.2	0.066
Pequea Creek	39,729	26.3	0.0	71.4	2.3	69.1	0.327	3.9	0.567
Pine Creek	255,207	88.5	0.6	10.5	0.4	72.6	0.225	13.0	0.021
Raystown Branch	186,222	64.6	0.6	33.6	1.2	73.9	0.233	9.8	0.070
Sherman Creek	63,334	69.2	0.3	30.3	0.2	73.7	0.218	9.8	0.081
Spring Creek	22,197	44.0	0.3	50.0	6.2	77.4	0.277	5.8	0.147
Swatara Creek	147,930	43.8	1.2	48.8	6.2	75.8	0.240	4.9	0.241
Tioga Creek	114,604	64.2	1.2	34.1	0.5	74.1	0.239	9.3	0.077
White Clay Creek	15,036	34.1	0.6	60.1	5.2	67.0	0.320	2.1	0.104
Wissahickon Creek	16,494	40.7	0.7	16.4	42.2	76.8	0.350	1.9	0.003
Young Woman Creek	11,981	99.7	0.0	0.3	0.0	71.9	0.190	12.5	0.000

---

[1] Includes mined/quarried areas and/or areas under development
[2] Measured in units of animal equivalent units (AEUs) per acre (see text for description)

Operating under the assumption that estimates of upland erosion provided by GWLF are reasonably accurate (at least for periods of one month or longer), monthly sediment loads for each watershed were calculated by starting with an initial "*a*" constant value of 6.66×10^{-4} and then varying this value until a match was achieved between simulated and observed loads. This initial "*a*" value is the annual value of 0.008 reported by Dietrich et al. (1999) divided by 12 in order to represent the monthly streambank-derived sediment loads being calculated by AVGWLF.

Figure 2 shows the calibration results for one watershed (Swatara Creek) aggregated on a seasonal basis. The top plot shows the match between observed and simulated data using the initial "a" value of 6.66×10^{-4} (which was too high), and the bottom one shows the match achieved with a final value of 7.99×10^{-5}. For the purposes of this study, a good match was assumed to be achieved when the simulated mean annual load (in kg/ha) was calculated to be equal to the observed mean annual load.

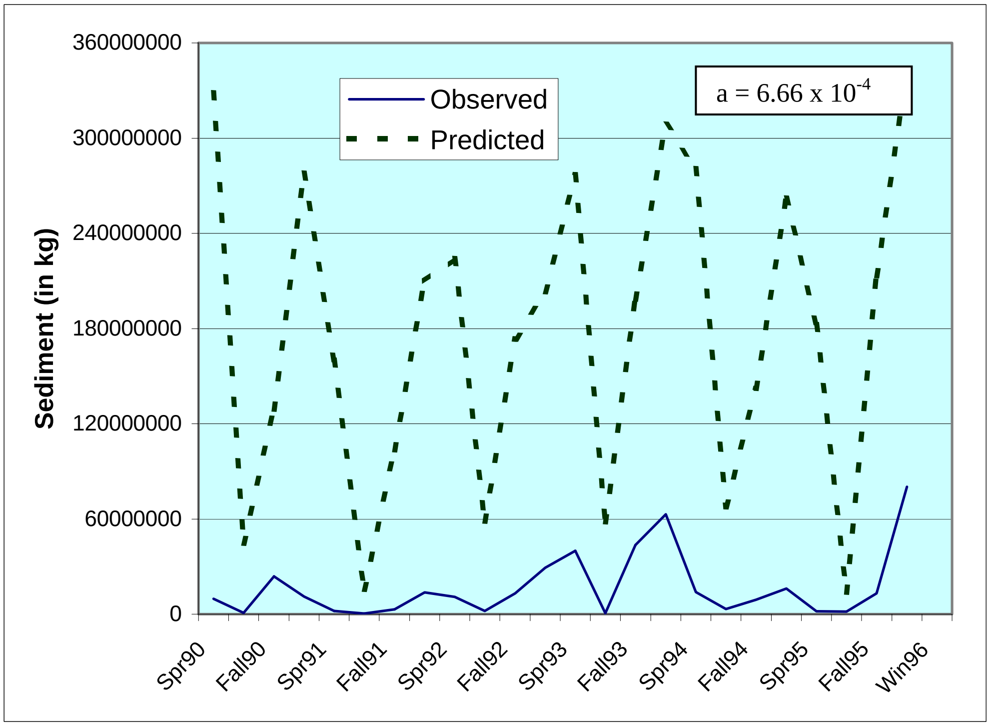
Once the "a" values for the watersheds had been estimated as described above, multiple linear regressions were then run between these values and a variety of watershed factors that could affect streambank erosion including mean curve number, mean land slope, percent developed area, farm animal density, mean soil erodibility (k factor), mean annual precipitation, stream density (meters/acre), watershed area, watershed perimeter, and watershed area divided by perimeter. Of these values, the best correlation ($r^2 = 0.69$) was found between "a" and mean watershed curve number, animal density, mean watershed k factor, and percent developed area. This was determined using the "best subsets" routine within the Minitab software package (Minitab Inc., 1996) which finds the best regression based on the maximum R-squared criterion. This routine first looks at all one-predictor models and selects the model giving the largest R-squared value. The largest R-squared value from all two-predictor models is then selected and so on until all potential model predictors are used. The best regression equation derived in this case was:

$$a = (0.00147 * PD) + (0.000143 * AD) - (0.000001 * CN) + (0.000425 * KF) + (0.000001 * MS) - 0.000016 \tag{3}$$

where : a = the empirical constant for calculating LER as described above,
PD = percent developed land in watershed,
AD = animal density measured in AEUs/acre,
CN = area-weighted curve number value of watershed,
KF = area-weighted k factor of the watershed,
MS = mean topographic slope (%) of the watershed.

In the above equation, animal density is expressed in animal equivalent units (AEUs) per acre, where one AEU is equal to 1000 pounds of animal weight. Curve numbers are empirically-derived values used in watershed hydrology simulation studies that reflect the relative amounts of surface runoff and infiltration occurring at a given location (U.S. Soil Conservation Service, 1986). These curve numbers are assigned on the basis of different combinations of soil and land use/cover type, and range in value from 1 to 100. The soil erodibility (k) factor is a measure of inherent soil erosion potential, and is primarily a function of soil texture, percent organic matter, soil structure, and permeability. Values typically are in units of tons/unit of rainfall erosion index for a 22 m-long overland flow length on a 9% slope in clean-tilled continuous fallow ground (Wischmeier, 1976).

Upon developing the regression model described above, an algorithm was constructed for automatically deriving the "a" factor within AVGWLF based on the use of GIS data sets currently used in the modeling system. A digital terrain (i.e., DEM) map is used to estimate average slope of



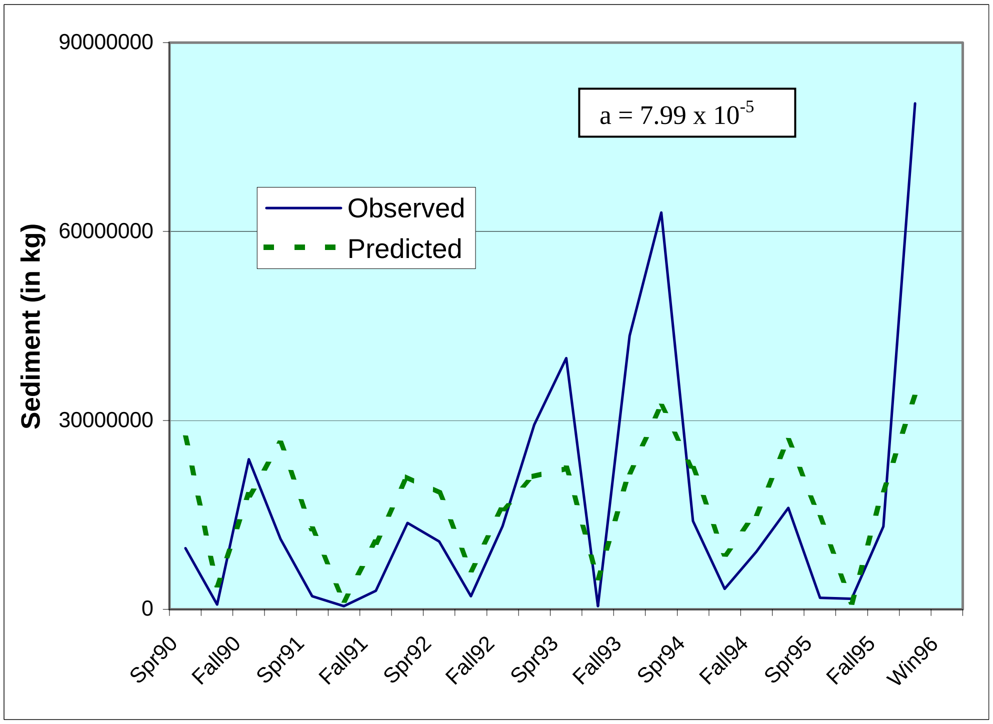


Figure 2. Comparison of seasonal sediment loads for Swatara Creek before (top) and after (bottom) adjustment of the “a” factor for calculating lateral erosion rate for the period April 1990 through March 1996.

the watershed, a digital soils map is used to estimate average USLE "k" factor, and a digital land use/cover map is used to estimate percent developed area. Both the soils map and the land use/cover map are used to estimate an average SCS curve number value for the watershed. Finally, a specially prepared "animal density" map based on farm animal populations recorded by postal zip code boundaries (see Evans, 2002) is used to predict animal density in a watershed.

As described earlier, within AVGWLF, equation (3) is used to first establish the "*a*" factor and subsequently calculate the monthly lateral erosion rate (LER) for a given watershed. Estimates of *Q* are derived from the monthly calculations of stream flow made by GWLF based on daily rainfall input. The estimated LER value is then multiplied by the total stream length of the watershed (in meters), a representative stream bank height (in meters), and a representative soil bulk density (in kg/m^3) to calculate monthly sediment loads generated by stream bank erosion. A digital stream map which depicts all "blue line" streams as shown on USGS 1:24,000-scale topographic maps is used to calculate total stream length. A mean stream bank height of 1.5 meters and a mean soil bulk density of 1500 kg/m^3 are used as representative default values within the model.

RESULTS AND DISCUSSION

To test the performance of the new stream bank erosion algorithm, sediment loads for the twenty-eight watersheds were re-calculated with the updated version of AVGWLF for the period 1989-1999. The simulated mean annual sediment loads were then compared with observed mean annual loads calculated using historical stream water quality data (see Table 2). A plot of the unit area sediment loads is shown in Figure 3. Simulated *a* values, lateral erosion rates, and sediment loads are shown in Table 3.

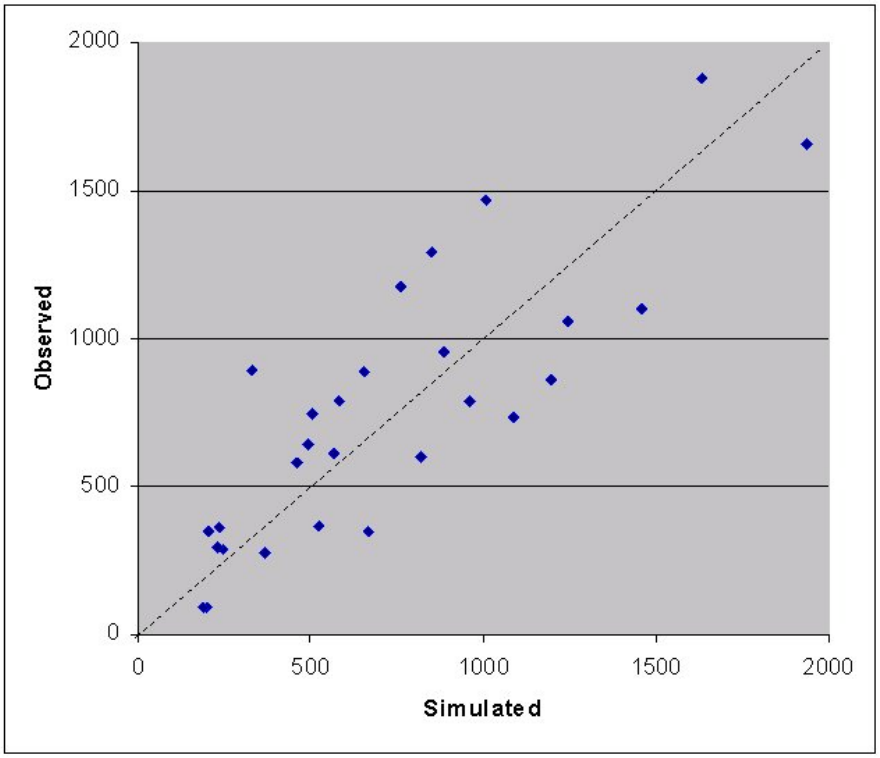


Figure 3. Observed versus simulated mean annual sediment loads (in kg/ha).

Table 2. Comparison of observed and predicted mean annual total sediment loads.

Watershed	Area (hectares)	Observed (Mg/yr)	Predicted (Mg/yr)	Observed (kg/ha)	Predicted (kg/ha)
Bald Eagle Creek	68,635	66,095.5	54,153.0	963	789
Brandywine Creek	74,908	63,821.6	96,706.2	852	1291
Brodhead Creek	77,864	38,231.2	50,066.6	491	643
Brokenstraw Creek	60,446	34,212.4	37,113.8	566	614
Bushkill Creek	30,426	6,085.2	2,860.0	200	94
Chartiers Creek	71,181	137,948.8	117,946.9	1938	1657
Chest Creek	81,620	88,884.2	60,153.9	1089	737
Conestoga River	122,969	200,439.5	231,181.7	1630	1880
Conodoguinet Creek	131,270	116,305.2	125,100.3	886	953
Driftwood Branch	77,094	18,194.2	27,830.9	236	361
Kettle Creek	63,810	15,824.9	18,377.3	248	288
Kishacoquillas Creek	48,340	15,378.5	17,692.4	525	366
Lehigh River	228,447	115,137.3	169,964.6	504	744
Loyalsock Creek	113,146	25,910.4	33,264.9	229	294
Lycoming Creek	55,805	11,495.8	19,308.5	206	346
Neshaminy Creek	53,940	54,263.6	79,130.0	1006	1467
Oil Creek	84,536	49,284.5	66,529.8	583	787
Paxton Creek	7,097	2,363.3	6,358.9	333	896
Penns Creek	80,353	53,515.1	28,203.0	666	351
Pequea Creek	39,729	57,924.9	43,622.4	1458	1098
Pine Creek	255,207	60,739.3	92,384.9	238	362
Raystown Branch	186,222	121,789.2	164,620.2	654	884
Sherman Creek	63,334	23,306.9	17,353.5	368	274
Spring Creek	22,197	26,592.0	19,133.8	1198	862
Swatara Creek	147,930	112,722.7	173,669.8	762	1174
Tioga River	114,604	52,717.8	66,355.7	460	579
White Clay Creek	15,036	12,329	9,006.6	820	599
Wissahickon Creek	16,494	20,485.5	17,434.2	1242	1057
Young Woman Creek	11,981	2,288.4	1,126.2	191	94

Table 3. Simulated erosion rates and sediment loads.

Watershed	Calculated "a" Value	Annual LER (m/yr)	Streambank (Mg/yr)[1]	Total (Mg/yr)[2]	Streambank (%)[3]
Bald Eagle Creek	8.0×10^{-5}	4.3×10^{-3}	4,450.8	54,153.0	8.2
Brandywine Creek	1.7×10^{-4}	1.0×10^{-2}	18,184.4	96,706.2	18.8
Brodhead Creek	8.1×10^{-5}	5.8×10^{-3}	9,977.1	50,066.6	19.9
Brokenstraw Creek	6.4×10^{-5}	4.8×10^{-3}	8,679.4	37,113.8	23.4
Bushkill Creek	2.1×10^{-5}	1.3×10^{-3}	727.4	2,860.0	25.4
Chartiers Creek	3.2×10^{-4}	1.4×10^{-2}	26,907.6	117,946.9	22.8
Chest Creek	5.3×10^{-5}	3.2×10^{-3}	7,197.9	60,153.9	12.0
Conestoga River	3.0×10^{-4}	1.8×10^{-2}	40,605.3	231,181.7	17.6
Conodoguinet Creek	1.2×10^{-4}	5.9×10^{-3}	17,778.0	125,100.3	14.2
Driftwood Branch	2.5×10^{-5}	1.7×10^{-3}	3,806.9	27,830.9	13.7
Kettle Creek	1.4×10^{-5}	9.7×10^{-4}	1,510.7	18,377.3	8.2
Kishacoquillas Creek	6.9×10^{-5}	2.2×10^{-3}	2,247.0	17,692.4	12.7
Lehigh River	4.7×10^{-5}	2.8×10^{-3}	13,637.9	169,964.6	8.0
Loyalsock Creek	2.3×10^{-5}	1.2×10^{-3}	3,399.5	33,264.9	10.1
Lycoming Creek	2.7×10^{-5}	1.4×10^{-3}	2,046.5	19,308.5	10.6
Neshaminy Creek	3.6×10^{-4}	1.8×10^{-2}	25,141.4	79,130.0	31.8
Oil Creek	6.8×10^{-5}	3.5×10^{-3}	8,998.6	66,529.8	13.5
Paxton Creek	5.8×10^{-4}	2.8×10^{-2}	5,001.1	6,358.9	78.6
Penns Creek	1.9×10^{-5}	1.0×10^{-3}	1,793.2	28,203.0	6.4
Pequea Creek	1.7×10^{-4}	8.5×10^{-3}	7,233.5	43,622.4	16.6
Pine Creek	1.6×10^{-5}	8.1×10^{-4}	4,750.5	92,384.9	5.1
Raystown Branch	3.7×10^{-5}	2.5×10^{-3}	15,386.4	164,620.2	9.3
Sherman Creek	1.8×10^{-5}	8.2×10^{-4}	1,389.0	17,353.5	8.0
Spring Creek	1.4×10^{-4}	7.0×10^{-3}	2,999.4	19,133.8	15.7
Swatara Creek	1.4×10^{-4}	7.6×10^{-3}	25,023.4	173,669.8	14.4
Tioga River	3.0×10^{-5}	1.3×10^{-3}	3,208.9	66,355.7	4.8
White Clay Creek	1.4×10^{-4}	4.2×10^{-3}	1,450.7	9,006.6	16.1
Wissahickon Creek	6.8×10^{-4}	3.3×10^{-2}	10,774.9	17,434.2	61.8
Young Woman Creek	1.0×10^{-5}	5.3×10^{-4}	129.7	1,126.2	11.5

[1] Sediment load contributed by streambank erosion on an annual basis
[2] Average annual sediment load from upland erosion plus streambank erosion
[3] Fraction of total sediment load contributed by streambank erosion

A number of statistical measures as suggested by Fitz et al. (2002) were used to evaluate model results, including model bias, root mean square error (RMSE), Pearson product-moment correlation coefficient (R^2), and the Nash-Sutcliffe coefficient. Brief overviews of these statistical measures are provided below.

Bias

Bias is calculated as follows:

$$\frac{\sum (y-x)}{n} \tag{4}$$

where *x* is the observed value, *y* is the model-simulated value, and *n* is the number of observations. As can be seen from this equation, bias is calculated as the mean differences between paired observed and simulated values. Bias values closer to zero indicate better overall model performance.

Root Mean Square Error (RMSE)

RMSE is calculated as:

$$\sqrt{\frac{\sum (y-x)^2}{n-1}} \tag{5}$$

where *x* is the observed value and *y* is the predicted value. As shown, RMSE is the square root of the average values of the prediction errors squared. This statistic is used to measure the discrepancy between modeled and observed values on an individual basis, and indicates the overall predictive accuracy of a model. Due to the quadratic term, greater weight is given to larger discrepancies. With this measure, smaller values indicate better model performance.

Pearson Product-Moment Correlation Coefficient (R^2)

This statistic is calculated as:

$$R^2 = \left(\frac{\sum (y-y_m)(x-x_m)}{\sqrt{\sum (y-y_m)^2 \sum (x-x_m)^2}} \right)^2 \tag{6}$$

where x_m is the mean of the observed (*x*) values, and *y* is the model-simulated value. The R^2 value is a measure of the degree of linear association between two variables, and represents the amount of variability that is explained by another variable (in this case, the model-simulated values). Depending on the strength of the linear relationship, the R^2 can vary from 0 to 1, with 1 indicating a perfect fit between observed and predicted values.

Nash-Sutcliffe Coefficient

The Nash-Sutcliffe coefficient is calculated as:

$$1 - \frac{\sum (y - x)^2}{\sum (x - x_m)^2} \tag{7}$$

where x_m is the mean of the observed data, and y is the model-simulated value. Like the R^2 measure described above, it is another indicator of "goodness of fit", and is one that has been recommended by the American Society of Civil Engineers (ASCE, 1993) for use in hydrological studies. With this coefficient, values equal to 1 indicate a perfect fit between observed and predicted data, and values equal to 0 indicate that the model is predicting no better than using the average of the observed data. Therefore, any positive value above 0 suggests that the model has some utility, with higher values indicating better model performance.

Values for the four statistical measures described above were calculated using the model results shown earlier in Table 2 (see Table 4). Based on these values, it appeared that the streambank erosion algorithm was performing reasonably well. As shown in Table 3, the estimated annual LER values ranged from a maximum of 3.3×10^{-2} m/yr to a minimum of 5.3×10^{-4} m/yr. For comparison purposes, it has been estimated that bare, unprotected stream banks can have annual lateral erosion rates of 1×10^{-2} to 2×10^{-1} m/yr (Prosser, et al., 2000; Rutherford, 2001; and Green et al., 1999). The latter values represent estimates for isolated "problem points" along a stream, whereas the values calculated in Table 3 represent average rates for all streams within a given watershed. The average values in the table are similar to the average lateral erosion rates of approximately 1×10^{-5} to 1×10^{-3} m/yr as reported by Morisawa (1969) for streams around the world.

Table 4. Calculated values for various statistical measures.

Statistical Measure	**Value**
Bias	34 kg/ha
Root Mean Square Error (RMSE)	257 kg/ha
Correlation Coefficient (R^2)	0.70
Nash-Sutcliffe Coefficient	0.68

From Table 3, it can also be seen that the stream bank-derived fraction of total sediment load varied from about 5% to 79%. Based upon a comprehensive literature review, Sekely et al. (2002) concluded that stream bank erosion contributes from about 17% to 92% of the total suspended sediment load of a stream depending upon conditions within the watershed. In this study, highly urbanized watersheds (e.g., Paxton Creek, and Wissahickon Creek) tended to have higher fractions from stream bank erosion. This coincides with studies described by Olem and Novotny (1994). The stream bank fraction tended to be proportionately lower in watersheds with greater percentages of agricultural land (e.g., Brandywine Creek, Conestoga River, Conodoguinet Creek, Spring Creek, Swatara Creek, and White Clay Creek) owing to the greater amounts of upland erosion in these areas. However, the stream bank fraction was again relatively high in agricultural watersheds that were also either fairly urbanized or had high grazing animal densities (e.g., Brokenstraw Creek, Chartiers, and Neshaminy Creek). As reported by Trimble (1994) and Williamson et al. (1992), stream bank erosion problems related to grazing can be quite significant in agricultural areas. Not surprisingly, both the upland erosion and stream bank fractions tended to be lower in watersheds that were primarily forested. As evidenced by the results in Table 3, the total mean annual loading rates in forested watersheds tend to be much lower than those found in watersheds where the landscape has been extensively altered by humans.

SUMMARY AND CONCLUSIONS

A GIS-based technique was developed for estimating streambank erosion rates for more accurately predicting total sediment loads at the watershed scale without the use of detailed field data. This technique relies on the use of data sets that are easily obtained and expressed as GIS data layers. This technique is based on empirically-derived relationships between “lateral erosion rates” and watershed characteristics such as curve number, grazing animal density, topographic slope, soil erodibility, and degree of urban development. An algorithm for estimating streambank erosion based on a statistical regression that reflected these relationships was incorporated into a GIS-based watershed model.

Simulated and observed sediment loads were compared for twenty-eight watersheds in Pennsylvania, and values obtained using four different statistical measures suggest that the model performed very well. In addition, the estimated lateral erosion rates seemed to be reasonable given similar rates reported by others in the literature. Finally, stream bank-derived fractions generally coincided with estimates derived by other researchers in watersheds having similar characteristics to those used in the study reported.

LITERATURE CITED

ASCE Task Committee on Definition of Criteria for Evaluation of Watershed Models of the Watershed Management Committee, Irrigation and Drainage Division, 1993. Criteria for evaluation of watershed models. Journal of Irrigation and Drainage Engineering, Vol. 199, No. 3.

Dietrich, C.R., T.R. Green, and A.J. Jakeman, 1999. An analytical model for stream sediment transport: application to Murray and Murrumbidgee river reaches, Australia. Hydrological Processes, Vol. 13, pp. 763-776.

Evans, B.M., D.W. Lehning, K.J. Corradini, G.W. Petersen, E. Nizeyimana, J.M. Hamlett, P.D. Robillard, and R.L. Day, 2002. A Comprehensive GIS-Based Modeling Approach for Predicting Nutrient Loads in Watersheds. J. Spatial Hydrology, Vol. 2, No. 2.

Fitz, H.C., N. Wang, J. Godin, F.H. Sklar, B. Trimble, and K. Rutchey, 2002. Calibration Performance of ELM v2.1a: 1979-1995 Water Quality and Hydrology. Report to the RECOVER Model Refinement Team, available at http://www.sfwmd.gov/org/wrp/elm/results/cal_ver/elm2.1/ELMcalibAnalysis_draft.pdf, South Florida Water Management District, West Palm Beach, FL.

Green, T.R., S.G. Beavis, C.R. Dietrich, and A.J. Jakeman, 1999. Relating stream-bank erosion to in-stream transport of suspended sediment. Hydrological Processes, Vol. 13, pp.777-787.

Haith, D.A. and L.L. Shoemaker, 1987. Generalized Watershed Loading Functions for Stream Flow Nutrients. Water Resources Bulletin, 23(3), pp. 471-478.

Haith, D.R., R. Mandel, and R.S. Wu, 1992. GWLF: Generalized Watershed Loading Functions User’s Manual, Vers. 2.0. Cornell University, Ithaca, NY.

Lemke, K.A., 1991. Transfer function models of suspended sediment concentrations. Water Resources Research, Vol. 27, No. 3, pp. 293-305.

Minitab Inc., 1996. Minitab User's Guide: Release for Windows^{TM}.

Morisawa, M., 1969. Streams: Their Dynamics and Morphology. McGraw-Hill Book Co., 175 pp.

Nelson, E.J. and D.B. Booth, 2002. Sediment sources in an urbanizing, mixed land-use watershed. Journal of Hydrology, 264, pp. 51-68.

Novotny, V. and H. Olem, 1994. Water Quality: Prevention, Identification, and Management of Diffuse Pollution. Van Nostrand Reinhold, New York.

Prosser, I.P., A.O Hughes, and I.D. Rutherford, 2000. Bank Erosion of an Incised Upland Channel by Subaerial Processes: Tasmania, Australia. Earth Surface Processes, Vol. 25, pp. 1085-1101.

Prosser, I., P. Rustomji, B. Young, C.Moran, and A. Hughes, 2001. Constructing River Basin Sediment Budgets for the National Land and Water Resources Audit. CSIRO Technical Report No. 15, 34 pp.

Rutherfurd, I. 2000. Some human impacts on Australian stream channel morphology. Brizga, S. and Finlayson, B. River Management: The Australasian Experience. Chichester, John Wiley & Sons.

Sekely, A.C., D.J. Mulla, and D.W. Bauer, 2002. Streambank slumping and its contribution to the phosphorus and suspended sediment loads of the Blue Earth River, Minnesota. J. Soil and Water Conservation, 57(5): 243-250.

Trimble, S.W., 1994. Erosional effects of cattle on streambanks in Tennessee, USA. Earth Surface Processes and Landforms, 19(5): 451-464.

U.S. Environmental Protection Agency, 2000. Total Maximum Daily Load (TMDL) Program. Rep. EPA841-F-00-009, Washington, DC.

Soil Conservation Service, 1986. Urban hydrology for small watersheds. Technical Release No. 55 (2nd edition). U.S. Department of Agriculture, Washington, DC.

VanSickle, J. and R.L. Beschta, 1983. Supply-Based Models of Suspended Sediment Transport in Streams. Water Resources Research, Vol. 19, No. 3, pp. 768-778.

Williamson, R.B., R.K. Smith, and J.M. Quinn, 1992. Effects of riparian grazing and channelization on streams in Southland, New Zealand. New Zealand Journal of Marine and Freshwater Research, 26(2): 241-258