A Comprehensive Modeling Approach for Reservoir Water Quality Assessment and Management Due to Point and Nonpoint Source Pollution

B. Narasimhan, R. Srinivasan, S. T. Bednarz, M. R. Ernst, P. M. Allen

Abstract. *A comprehensive modeling approach has been developed for use in formulating a watershed management plan to improve the water quality of Cedar Creek reservoir, one of five large water supply reservoirs in north central Texas operated by Tarrant Regional Water District. Eutrophication, or specifically the increase in concentrations of chlorophyll-a (chl'a') over the last 18 years, is a major concern for the water managers. To develop a watershed management plan, the watershed model SWAT was linked with the lake eutrophication model WASP. Several intensive field campaigns and surveys were conducted to collect extensive water quality and land management data for model setup and calibration. In addition to the streamflow, the SWAT model was well calibrated for sediment (including channel erosion) and nutrients. Further, a simple modification to the SWAT in-stream routine allowed simulation of the nutrient load due to channel erosion. The in-stream water quality parameters for SWAT were based on an independent QUAL-2E model calibration. The calibrated SWAT model showed that more than 85% of the total N and total P loading to the lake are from watershed nonpoint sources. Although cropland occupies only 6% of the watershed area, it contributed more than 43% of the sediment, 23% of total N, and 42% of total P loading from the watershed. The channel erosion contributed about 35% of the total sediment load. The watershed model identified subbasins that contribute considerable amounts of sediment and nutrients. Based on these loads, the calibrated WASP model showed that the watershed nonpoint-source nutrient load (total N and total P) should be reduced by at least 35% to see a significant reduction in chl'a' concentrations when compared to the WASP calibration levels.*

Keywords. *Channel erosion, Eutrophication, Hydrologic modeling, QUAL-2E, Reservoir water quality, SWAT, WASP, Watershed management plan.*

According to the U.S. EPA, eutrophication accounts for more than 50% of the impaired lake area and 60% of the impaired river reaches in the country (USEPA, 1996). Eutrophication is the process in which excessive algal growth takes place due to the accumulation of surplus nutrients, primarily nitrogen and phosphorus, in slow-moving water bodies. Algal blooms increase the possibility of fish kills from ammonia toxicity in the afternoon and hypoxic conditions before dawn. Further, eutrophication also causes substantial economic losses such as high drinking water treatment costs, reduced recreational value, and reduced property values (Dodds et al., 2009).

Submitted for review in November 2009 as manuscript number SW 8308; approved for publication by the Soil & Water Division of ASABE in June 2010.

The authors are **Balaji Narasimhan, ASABE Member,** Assistant Professor, Department of Civil Engineering, Indian Institute of Technology Madras, India; **Raghavan Srinivasan, ASABE Member,** Director and Professor, Spatial Sciences Laboratory, Texas A&M University, College Station, Texas; **Steven T. Bednarz, ASABE Member,** Civil Engineer, USDA-NRCS, Temple, Texas; **Mark R. Ernst,** Water Quality Manager, Tarrant Regional Water District, Fort Worth, Texas; and **Peter M. Allen, ASABE Member,** Professor, Department of Geology, Baylor University, Waco, Texas. **Corresponding author:** Balaji Narasimhan, EWRE-29, Department of Civil Engineering, Indian Institute of Technology Madras, Chennai 600036, Tamil Nadu, India; phone: +91-44-22574293; fax: +91-44-22574252; e-mail: nbalaji@iitm.ac.in.

Excessive nutrients to a water body can come from a variety of sources, such as nutrient-enriched runoff from agricultural fields, lawns, golf courses, and discharges from wastewater treatment plants. In order to prevent the hypoxic condition and restore the water quality, total maximum daily loads (TMDLs) are being developed for many nutrient-impaired water bodies. Extensive water quality observations together with spatially distributed watershed models such as Soil and Water Assessment Tool (SWAT) and Hydrological Simulation Program-Fortran (HSPF) have been extensively used for developing TMDLs by simulating the hydrology, sediment, nutrient, and pollutant loading of large basins.

Watershed models such as SWAT and HSPF are quite effective at capturing the spatial variability of a watershed in terms of topography, landuse, soil, and management practices. This is because overland hydrologic processes (runoff, infiltration, and evapotranspiration), soil erosion, and nutrient/pesticide wash-off and leaching processes are represented in sufficient detail in these watershed models. However, the biogeochemical transformations of nutrient and pollutant loadings once they reach the streams and reservoirs are simulated in a simplified way by assuming them to be a completely mixed system. Conversely, in water quality models such as Water Analysis Simulation Program (WASP) and CE-QUAL-W2, the biogeochemical processes are represented in sufficient detail, but the overland hydrologic and transport processes are not at all considered. Hence, it is necessary to link hydrologic models with reservoir water quality models to comprehensively assess the impact of overland processes on the quality of receiving water bodies.

Both HSPF and SWAT have been linked in the past with lake water quality models. One of the earliest linkages between watershed and a receiving waterbody models was developed for the Southwest Florida Water Management District (Wool et al., 1994). Wool et al. (1994) developed the Linked Watershed/Waterbody Model (LWWM) to interface EPA's Stormwater Management Model (SWMM) with WASP to rapidly evaluate the effect of point and nonpoint sources on the quality of receiving waters. The nonpoint source loading simulations of the SWMM RUNOFF block were mapped to WASP segments for simulating the resultant water quality. LWWM is used by the Southwest Florida Water Management District to study the impact of watershed development and for developing best management practices (BMPs) to preserve the water quality (www.swfwmd.state.fl. us).

Wu et al. (2006) linked HSPF with the hydrodynamic water quality model (CE-QUAL-W2) for developing BMPs to control nonpoint source pollutant loading from the Swift Creek reservoir watershed in Virginia. In a similar study at the Occoquan watershed in Virginia, six HSPF models and two CE-QUAL-W2 models have been complexly linked to simulate two major reservoirs and the associated drainage areas in a comprehensive way (Xu et al., 2007). The hydrologic and water quality integration tool (HydroWAMIT), developed by combining features of HSPF and GWLF (Generalized Watershed Loading Functions), has been linked with WASP and applied to the north and south branches of the Raritan River watershed in New Jersey (Cerucci and Jaligama, 2008).

Recently SWAT has also been linked with CE-QUAL-W2 (Debele et al., 2006) for Cedar Creek reservoir in north central Texas. In the current study, a comprehensive modeling approach has been developed by linking SWAT with WASP to understand the impact of point and nonpoint source loading from the watershed on the lake water quality. Such a linkage would be of immense help during the TMDL process to spatially identify and quantify the sources of sediment and nutrient loading into the lake and design appropriate remedial measures for improving the water quality.

Problem Statement

Cedar Creek reservoir (fig. 1), with a surface area of 13,350 ha and a volume of 795 million m^3, is one of five major reservoirs supplying water to about 1.7 million people across 11 counties (65 cities including the city of Fort Worth) in north central Texas and is maintained by Tarrant Regional Water District (TRWD). By 2050, these reservoirs are ex-

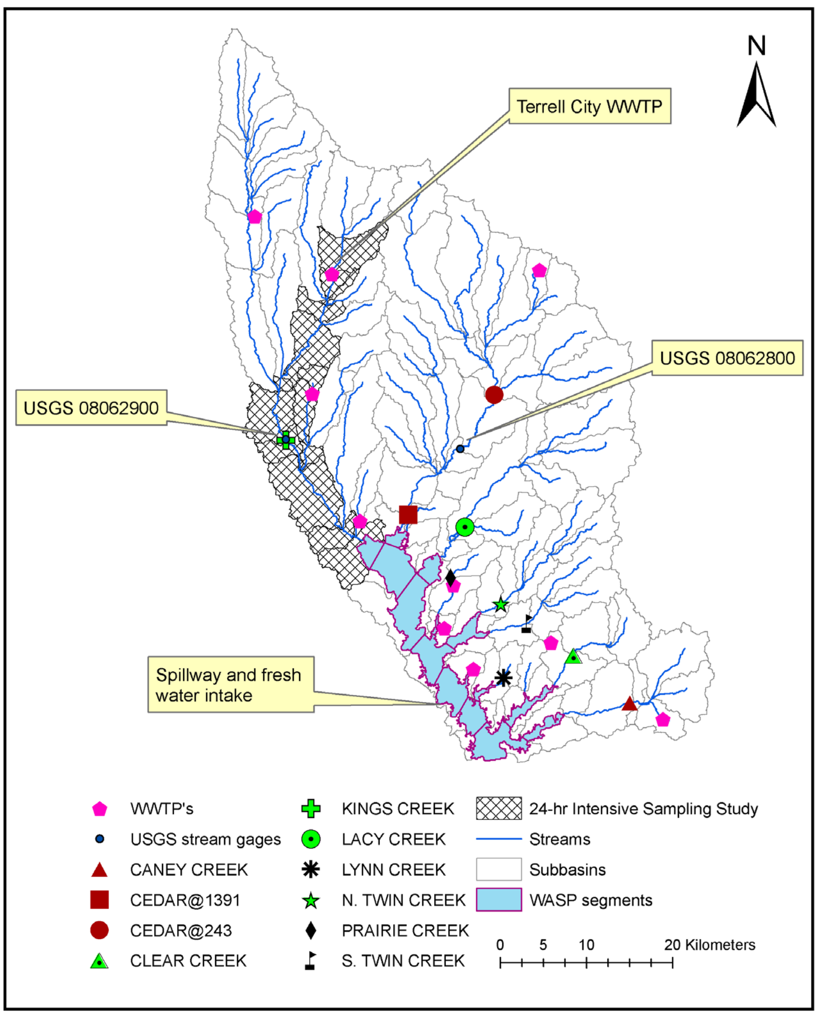

Figure 1. Cedar Creek watershed.

pected to meet the water demand of over 2.66 million people. Hence, water quality in north central Texas reservoirs is a growing concern due to rapid urbanization and changing land management practices in the watershed. Cedar Creek has already been listed as impaired (303 d) for high pH violations resulting from excessive photosynthesis and low alkalinities. Currently it ranks 11th among 104 Texas reservoirs for highest chlorophyll-*a* (chl'*a*') (Ernst and Owens, 2009). Based on 18 years of data, the third quarter (July to September) median concentration of chl'*a*' is about 27.4 $\mu g\ L^{-1}$. Long-term monitoring of water quality in Cedar Creek reservoir showed that since 1989 chl'*a*' has been increasing annually at a rate of 3.85% (Ernst and Owens, 2009). This increase in chl'*a*' could be due to changes in land management practices, urbanization, and point source loads from wastewater treatment plants. Sedimentation from overland and channel erosion is another major concern as it affects water quality and reduces reservoir storage capacity. Further, sediments also carry significant quantities of nutrients that stimulate algal growth, causing eutrophication, or the increase in chl'*a*' concentration.

The objective of the current study is to develop a comprehensive modeling approach by linking a watershed model with a lake water quality model to spatially identify and quantify the sources of sediment and nutrient loading to Cedar Creek reservoir in conjunction with field observations and surveys. As the landuse of the watershed is primarily rural (<7% urban), the hydrologic model SWAT was used due to its ability to simulate crop growth and complex land management practices. SWAT was linked with the three-dimensional lake water quality model WASP to simulate the resultant algal growth (chl'*a*') due to nutrient loadings simulated from the watershed. The modeling approach could be used in formulating a watershed management plan to improve the water quality of Cedar Creek reservoir.

Methodology

Watershed Model Setup

The SWAT model (SWAT2000) integrated with the EPA BASINS 3.0 interface was used in this study for watershed simulation. Many changes have been made to the water and sediment routing routines of the SWAT2000 code, as described by Narasimhan et al. (2007) and used for this study. These changes have been subsequently incorporated into SWAT2005 and SWAT2009. Topography data needed for watershed delineation were obtained from the 1:24,000-scale USGS National Elevation Database. At the time of initiation of this project in 2001, NLCD 2001 landuse data were not available. Hence, landuse data for the watershed were obtained from the 1992 National Land Cover Dataset (NLCD). However, to account for urbanization, the extent of urban landuse was updated in the 1992 landuse map using 2001 LANDSAT 7 ETM+ satellite imagery. The detailed county-level soil database (SSURGO) was obtained from the USDA Natural Resource Conservation Service (NRCS). The watershed was divided into 106 subbasins based on the natural topography of the region. These 106 subbasins were further subdivided into 1,516 hydrologic response units (HRUs) based on unique combinations of soil and landuse.

Cedar Creek watershed contains about 120 inventory-sized dams (as defined by the Texas Commission on Environmental Quality), which include NRCS flood prevention dams, farm ponds, and other privately owned dams. The physical data (e.g., surface area, storage, drainage area, discharge rates) for these dams were input to the SWAT model to allow routing of runoff through the structures. Four structures were big enough (>1500 acre ft) to be simulated as reservoirs. The rest of the structures were simulated as small ponds.

Based on the crop statistics from the NRCS field offices, sorghum is the dominant crop grown in the cropland of this watershed. The majority of the cropland is cultivated using conventional tillage with a typical fertilization rate of 67 kg N and 34 kg P per hectare. Field survey showed that the pasturelands are in fair hydrologic condition, and at least 50% of them are fertilized (64 kg N) in any given year with at least two hay cuttings. The pervious portions of the urban land (lawns and golf courses) were assumed to be planted with bermudagrass. Because good estimates of urban fertilization rates are not available, the required amount of fertilizer was automatically applied by SWAT. The fertilizer was automatically applied whenever the growth rate of the plant undergoing nitrogen stress level fell 10% below the potential growth rate (AUTO_NSTR = 0.9) in the urban landscape.

In SWAT, by default, the concentrations of nutrients in the soil layers are initialized based on organic carbon content from the soils database by assuming a C:N ratio of 14:1 and an N:P ratio of 8:1. These ratios are typical for soils in natural conditions. However, in croplands, the nutrients (especially mineral phosphorus) tend to build up in the soil if fertilizer is routinely applied without proper soil testing. Although the watershed is currently dominated by pastureland (60%) for hay production, historically until the 1980s almost all of this pastureland was used for row crop production, such as cotton, corn, or sorghum, with intensive fertilizer and soil management. Hence, the soil nutrients in these pasturelands will be much higher than natural conditions. This was confirmed based on limited analysis of soil samples as well. If a method based on C:N ratio alone were used to initialize the soil nutrients in pastureland, then the actual nutrient loading from pastureland and cropland would be underestimated.

In order to initialize the soil nutrient values, all pasturelands were assumed to be managed in the same way as croplands, and the model was run for 37 years. The simulated soil nutrient values at the end of the 37 years of simulation were used as the initial soil nutrient value for all pasturelands and croplands for subsequent calibration of the model.

Cedar Creek watershed contains nine wastewater treatment plants (WWTPs) distributed across the watershed, and two of these WWTPs discharge directly into Cedar Creek reservoir (fig. 1). Wastewater treatment plant loading was estimated based on one year of weekly nutrient and flow data voluntarily collected and measured by the WWTPs themselves. These weekly data were cumulated into monthly loadings for each WWTP and routed through the creeks. The same data for one year were used in the entire model run of 37 years.

Field surveys showed that channel erosion is rampant across the watershed. Hence, it is important to quantify channel erosion and model it accurately to partition the sediment load and the associated nutrient load coming from overland erosion. A rapid geomorphic field assessment, similar to the RAP-M method adopted by the NRCS (Windhorn, 2001), was done to estimate channel erosion (Allen et al., 2007).

Further, the annual reservoir sedimentation rate was calculated based on volumetric survey data and lake bottom sediment core analysis. The study by Allen et al. (2007) showed that the total annual sediment load to the Cedar Creek lake is about 446,558 metric tons per year (t year^{-1}). Based on the RAP-M method, the channel erosion contribution is estimated to be about 152,572 t year^{-1}. From the difference between the total annual sediment load and channel erosion, the overland erosion could be inferred as 293,986 t year^{-1} (Allen et al., 2007). This information was used to calibrate the SWAT parameters for overland and channel erosion.

In order to accurately predict channel erosion, a few modifications were made to the SWAT2000 code. SWAT predicts channel erosion as a function of flow velocity using a simplified stream power approach developed by Bagnold (1977) and adopted by Williams (1980). Hence, accurate prediction of flow velocity is very important to simulate channel degradation and deposition. SWAT uses a bucket-type approach for calculating the amount of water entering the reach, i.e., the water is moved from one reach to another by volume basis. The flow depth is calculated by dividing the volume of water entering the reach during that day by the length of the reach, which is subsequently used to calculate the flow velocity. This approach overestimates the flow depth, and hence the velocity, in a smaller reach downstream of the confluence of two bigger reaches. This is because the large volume of water entering the reach is forced to fit within the smaller reach length. Hence, a simple iterative approach, as described by Narasimhan et al. (2007), was used to calculate the flow depth until the flow rate of the channel is equal to the daily flow rate of water entering the reach. In addition to this, a few other modifications were also made to the SWAT2000 code to explicitly output the quantity of channel erosion predicted every day.

As stated before, channel erosion was found to contribute about 35% of total sediment loading in the watershed (Allen et al., 2007). These sediments also contribute a substantial amount of organic nutrient loading into the lake, which should be accounted for while quantifying the nutrient loads from various sources. Currently, SWAT does not account for nutrients contributed by channel erosion. Hence, the in-stream water quality routine of SWAT was modified to simulate the nutrient loading in proportion to the amount of channel erosion based on the measured nutrient concentration in the stream bank. The nutrient loading due to channel erosion is calculated as:

$$ch\_nut_{rch} = ch\_erosion_{rch} \times ch\_nutco_{rch} / 1000 \quad (1)$$

where ch_nut_{rch} is nutrient loading (NO_3, labile P, organic N, and organic P) from the channel (*rch*) due to channel bank erosion (kg) during the time step; $ch\_erosion_{rch}$ is bank erosion from the channel (metric tons); and ch_nutco_{rch} is the concentration of the nutrients (NO_3, labile P, organic N, and organic P) in the channel bank sediment (mg kg^{-1} or ppm). The nutrient loading from the bank erosion is added to the appropriate pools of nutrients coming from overland flow and upstream reaches and routed using the QUAL-2E in-stream processes embedded in SWAT. These changes have been subsequently incorporated into SWAT2005.

For the purpose of this study, nine soil samples from the channel banks were collected across the watershed and analyzed for nitrate and phosphate. Based on the laboratory analysis, constant values of 1.89 ppm of nitrate and 7.2 ppm of labile phosphate were used for stream bank sediments across the watershed. The organic nitrogen (orgN) values of stream bank sediments were derived from the organic carbon concentrations from SSURGO soil polygons adjacent to the streams by assuming a C:N ratio of 14:1 for humic materials. The organic phosphorus (orgP) concentrations in stream bank sediments were estimated by assuming an orgN:orgP ratio of 8:1 for humic materials.

Water Quality Model Setup

QUAL-2E

Detailed studies of flow and travel time were done during various flow conditions in order to accurately set up the QUAL-2E in-stream hydraulics. The travel time was measured by conducting a fluorometric dye (Rhodamine) study and analyzing the breakthrough curves at various points along the stream. An independent QUAL-2E model was set up based on the measured channel geometry and hydraulics developed during the dye study. During a separate study, 24 h water quality samples were collected at various points along Kings Creek downstream of a major wastewater treatment plant (City of Terrell). The measured concentration of dissolved oxygen (DO), biological oxygen demand (BOD), ammonia, phosphorus, chl'*a*', organic nitrogen, and nitrate + nitrite (NOx) concentrations at various locations along Kings Creek were used as input data to populate the model's initial conditions and for calibration. The calibrated QUAL-2E kinetic terms and coefficients were used as initial estimates to set up the in-stream water quality parameters of SWAT. Further, water quality of grab samples collected periodically by TRWD from 1989 to 2002 at ten monitoring stations on the major tributaries to the Cedar Creek reservoir (fig. 1) were used to modify and calibrate SWAT's in-stream model parameters.

WASP

SWAT provided daily flow and loads for the WASP model to allow for an 11-year simulation. Cedar Creek was partitioned into 22 segments with up to three vertical segments in the main pool (Ernst and Owens, 2009). The segmentation in WASP was based on temperature stratifications along with physical characteristics of the lake such as depth and incoming tributary flows.

The nonpoint source loading simulated by SWAT along with the point source discharges from seven WWTPs located upstream and two WWTPs discharging directly to the lake were used as inputs to WASP. A generic program was written in the PERL scripting language to prepare the nonpoint source file that summarizes the flow and nutrient loads from different SWAT subbasins and reach segments to the corresponding WASP segments.

The Cedar Creek WASP model was calibrated for an 11-year period from 1991 to 2001. Calibration involved comparison of median observed data for main channel reservoir segments to median predicted data for the entire 11-year period. More details on the WASP model setup for Cedar Creek, calibration, and validation can be obtained from Ernst and Owens (2009).

Results and Discussion

Stream Flow Calibration and Validation

Measured stream flow was obtained from two USGS stream gages in the watershed from 1963 through 1987, and this period was used for initial calibration (fig. 1). SWAT was calibrated for flow by adjusting appropriate inputs that affect surface runoff and base flow (Santhi et al., 2001). Adjustments were made to runoff curve number, soil evaporation compensation factor, shallow aquifer storage, shallow aquifer re-evaporation, channel Manning's *n*, and channel transmission loss (table 1) until the simulated total flow and fraction of base flow were approximately equal to the measured total flow and base flow, respectively.

Validation was performed by comparing SWAT-simulated inflows to the reservoir with total reservoir inflows calculated from measured daily reservoir volume, water surface evaporation, withdrawals, discharges, and rainfall by a simple mass balance approach.

After a three-year model warm-up period, flow calibration was performed on the monthly flow data from 1966 through 1987. During this period, the predicted flow matched very well at the USGS stream gages 08062800 (Cedar Creek) and 08062900 (Kings Creek); R^2 values were 0.82 at Cedar Creek and 0.89 at Kings Creek (figs. 2a and 2b). Measured and predicted mean flows were also nearly equal. The Nash-Sutcliffe coefficients of efficiency (E; Nash and Sutcliffe, 1970) were 0.81 and 0.83, which also indicated a good calibration of the model.

With the same calibrated inputs, stream flow was validated from 1980 through 2002 using the measured mass balance of Cedar Creek reservoir for comparison to predicted inflow values (fig. 2c). The predicted inflow at this independent location, matched very well with the measured inflow, with an R^2 of 0.76 and Nash-Sutcliffe E of 0.79 (fig. 2c).

SWAT mean flow predictions were slightly higher than the measured mean flows. This anomaly is not necessarily due to the difference in high flow predictions but due to differences in the low flow prediction. Visual inspection of the hydrographs indicates that the peaks were reasonably well predicted. Few mismatches in the peak streamflow prediction are primarily due to the spatial variability of particular rainfall events that were not adequately captured by the existing rain gauges.

In order to better predict the low flows, a baseflow filter developed by Arnold et al. (1995) was used to quantify the fraction of baseflow contribution to the total streamflow based on the streamgage measurements. The same filter was used on the SWAT-predicted streamflow to find the baseflow fraction. SWAT parameters related to groundwater, listed in table 1, were adjusted until the baseflow fraction from SWAT was almost equal to those estimated from the streamgages. In spite of this, there is a slight tendency to overpredict the low flow events.

For accurate simulation of travel time through the stream network, the flow velocity simulated during the dry weather flow (only discharges from WWTPs) was converted to travel time based on the length of the channel and compared with the travel time estimates from the fluorometric dye study. Based on the analysis from the dye study, the estimated travel time from Terrell WWTP to the reservoir is about ten days. SWAT simulations, with the default Manning's *n* of 0.014, resulted in a travel time of about five days. In an effort to lengthen the travel time in SWAT, Manning's *n* was raised to 0.075 along with channel transmission losses (table 1). A value of 0.075 appeared to be more appropriate, as the channels were sluggish and woody with deep pools. With a Manning's *n* of 0.075, the travel time predicted by SWAT was about 9.5 days. Based on these parameters, the measured flow and SWAT-predicted flow during an intensive 24 h sampling study showed a good match (fig. 3). The flows predicted by SWAT matched the measured flows better than those of QUAL-2E because QUAL-2E uses a simple hydraulic equation based on power law relationships.

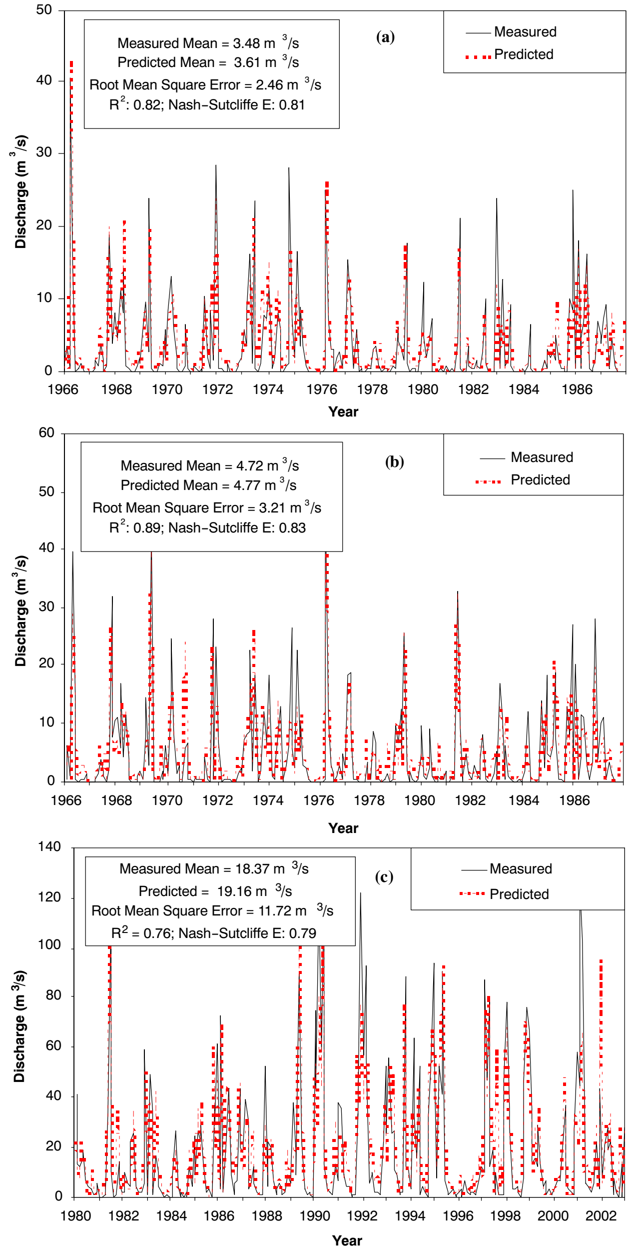

Figure 2. Calibration of stream flow at (a) Cedar Creek (b) Kings Creek and (c) Validation of total inflows to Cedar Creek reservoir.

Sediment Calibration

Annual overland and channel erosion rates determined from a watershed and lake survey (Allen et al., 2007) were used as target rates for adjusting the SWAT model parameters

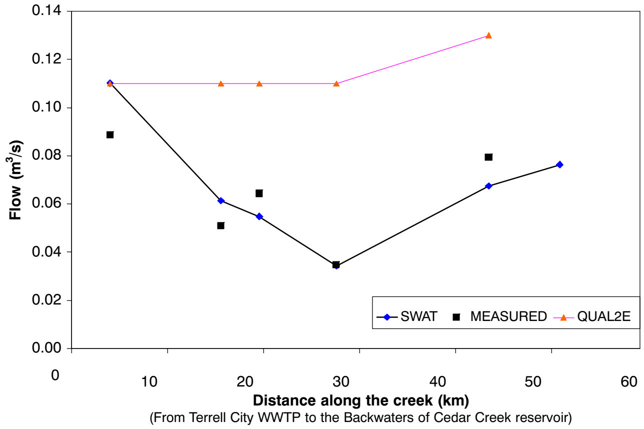

Figure 3. Calibration of low flow condition at stations on Kings Creek (24 h intensive sampling study in fig. 1).

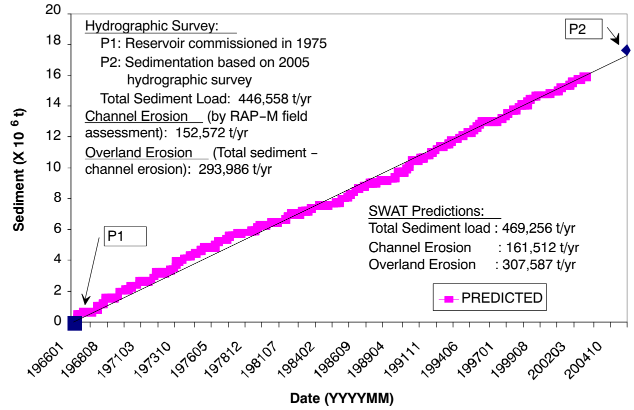

Figure 4. Channel and overland erosion in the watershed.

(table 1). The channel erosion power function parameters (SPCON and SPEXP) were adjusted based on limited storm flow total suspended solids (TSS) data available at various stream segments. The coefficients were chosen in such a way that the average simulated suspended sediment concentration is two to three times higher than the measured TSS at the ten periodic grab sample locations distributed across the creek (fig. 1). This was done because SWAT does not simulate bed load transport and all the sediments are assumed to be in suspension. If these coefficients (SPCON and SPEXP) were tightly calibrated with measured TSS without accounting for bed load transport, the model would considerably underestimate the sediment transport power of the water.

Channel physical properties, such as channel vegetation cover factor and channel erodibility factor, were adjusted for individual stream segments based on field assessment and geological data. The higher these values are, the greater is the vulnerability of the channel for erosion, and vice versa. As the pastureland was found to be in fair hydrologic condition, the USLE cover factor was reduced to 0.007. Based on these calibrated parameters, the simulated overland and channel erosion rates were within 5% of the erosion estimated from field survey (fig. 4).

Estimating annual sediment load into the lake is a challenging task if the watershed is not adequately instrumented to measure sediments. Further, for modeling as well as for developing best management practices, it becomes essential to partition the sediment load into channel erosion and overland erosion. However, the Cedar Creek watershed is not instrumented to monitor overland erosion or channel erosion. Hence, a field survey by adopting the rapid geomorphic assessment proved extremely useful to quantify channel ero–

Table 1. SWAT watershed coefficients adjusted for calibration of flow, sediment and nutrient.

Variable	Description	Input Value	Units	SWAT File
Coefficients related to flow				
CN2	SCS runoff curve number (adjustment range)	+3 to -3	--	*.mgt
ESCO	Soil evaporation factor	0.85	--	*.hru
GW_REVAP	Groundwater re-evaporation coefficient	0.10	--	*.gw
GW_DELAY	Groundwater delay time	135	days	*.gw
GWQMN	Groundwater storage required for return flow	1.00	mm	*.gw
REVAPMN	Groundwater storage required for re-evaporation	1.60	mm	*.gw
ALPHA_BF	Baseflow alpha factor	0.0420 to 0.2006	day^{-1}	*.gw
CH_N2	Manning's *n* roughness for channel flow	0.075	--	*.rte
CH_K2	Hydraulic conductivity of channel alluvium	0.1 to 4.0	mm h^{-1}	*.rte
Coefficients related to sediment				
RSDIN	Initial soil residue cover	1000	kg ha^{-1}	*.hru
USLE_C	Minimum C value for pastureland in fair condition	0.007	--	crop.dat
SPCON	Linear parameter for calculating the maximum amount of sediment that can be re-entrained during channel sediment routing	0.01	--	basins.bsn
SPEXP	Exponent parameter for calculating sediment re-entrained in channel sediment routing	1.4	--	basins.bsn
CH_COV	Channel cover factor	0.1 to 1.0	--	*.rte
CH_EROD	Channel erodibility factor	0.3 to 0.8	--	*.rte
Coefficients related to nutrient				
CMN	Rate factor for humus mineralization of active organic nitrogen	0.003	--	basins.bsn
UBN	Nitrogen uptake distribution parameter	20	--	basins.bsn
UBP	Phosphorus uptake distribution parameter	100	--	basins.bsn
NPERCO	Nitrogen percolation coefficient	0.2	--	basins.bsn
PPERCO	Phosphorus percolation coefficient	10	--	basins.bsn
PHOSKD	Phosphorus soil partitioning coefficient	200	--	basins.bsn
PSP	Phosphorus sorption coefficient	0.4	--	basins.bsn

Table 2. In-stream water quality coefficients for calibration of QUAL2E and SWAT.

Variable	Definition	QUAL-2E	SWAT
LAO	Light averaging option	2	2
IGROPT	Algal specific growth rate option	2	2
AI0	Ratio of chlorophyll-*a* to algal biomass (μg chl'*a*' mg^{-1} algae)	10	10
AI1	Fraction of algal biomass that is nitrogen (mg N mg^{-1} algae)	0.090	0.090
AI2	Fraction of algal biomass that is phosphorus (mg P mg^{-1} algae)	0.020	0.020
AI3	Rate of oxygen production per unit of algal photosynthesis (mg O_2 mg^{-1} algae)	1.600	1.400
AI4	Rate of oxygen uptake per unit of algal respiration (mg O_2 mg^{-1} algae)	2.300	2.000
AI5	Rate of oxygen uptake per unit of NH_3-N oxidation (mg O_2 mg^{-1} NH_3-N)	3.500	3.000
AI6	Rate of oxygen uptake per unit of NO_2-N oxidation (mg O_2 mg^{-1} NO_2-N)	1.000	1.000
MUMAX	Maximum specific algal growth rate at 20°C (day^{-1})	1.800	1.000
RHOQ	Algal respiration rate at 20°C (day^{-1})	0.100	0.300
TFACT	Fraction of solar radiation computed in the temperature heat balance that is photosynthetically active	0.300	0.300
K_L	Half-saturation coefficient for light (kJ m^{-2} min^{-1})	0.418	0.418
K_N	Michaelis-Menten half-saturation constant for nitrogen (mg N L^{-1})	0.400	0.400
K_P	Michaelis-Menten half-saturation constant for phosphorus (mg P L^{-1})	0.040	0.040
P_N	Algal preference factor for ammonia	0.100	0.100
RS1	Local algal settling (0.15 to 1.82) (default = 1.0)	0.1	0.01
RS2	Benthos source rate for dissolved P (default = 0.05)	0	0.001
RS3	Benthos source rate for NH_4-N (default = 0.5)	0	0.001
RS4	Organic N settling rate (0.001 to 0.10) (default = 0.05)	0.1	0.01
RS5	Organic P settling rate (0.001 to 0.10) (default = 0.05)	0.1	0.01
RK1	CBOD deoxygenation rate (0.02 to 3.4) (default = 1.71)	0.055	0.01-0.050
RK2	Reaeration rate (0.01 to 100) (default = 50.0)	1.17 - 15.89	0.5 - 1.5
RK3	CBOD settling loss rate (-0.36 to 0.36) (default = 0.36)	0.01 - 0.1	0.025 - 0.25
RK4	Benthic oxygen demand (default = 2.0)	0.8	0.8
BC1	Decay rate for NH_4 to NO_2 (0.1 to 1.0) (default = 0.55)	0.2 - 0.6	0.3
BC2	Decay rate for NO_2 to NO_3 (0.2 to 2.0) (default = 1.1)	0.08 - 0.15	1.2
BC3	Decay rate for organic N to NH_4 (0.2 to 0.4) (default = 0.21)	0.001 - 0.1	0.03 - 0.2
BC4	Decay rate for organic P to dissolved P (0.01 to 0.70) (default = 0.35)	0.05	0.01

sion contribution to the total sediment load. However, such an estimate of channel erosion from rapid field assessment has considerable uncertainty associated with the estimates, because the assessment is qualitative rather than quantitative. The uncertainty arises mainly due to properly categorizing the channel into one of the channel evolution models based on down cutting and widening due to erosion and in assessing the erosion rates. In order to reduce the uncertainty, erosion pins could be installed at several channel cross-sections to get a better estimate of erosion rates. Nevertheless, the rapid field assessment was able to provide a first-cut estimate of annual channel erosion (152,572 t $year^{-1}$), based on which the overland sediment load (293,986 t $year^{-1}$) could be inferred given the average annual sediment load (446,558 t $year^{-1}$) to the lake from the lake hydrographic survey.

In-Stream Water Quality Calibration

The in-stream water quality calibration was done in two stages: (1) based on an intensive 24 h sampling study conducted downstream of the Terrell WWTP during the low flow in Kings Creek segment, and (2) based on long-term, periodic grab samples collected at ten monitoring locations at various tributaries draining into the Cedar Creek reservoir (fig. 1).

An independent QUAL-2E model was calibrated based on the measured water quality parameters during the 24 h sampling study at specified location along the creek downstream of Terrell WWTP. The kinetic rates, reaeration rates, and settling rates of various constituents such as BOD, algae, and nutrients were adjusted during calibration (table 2). These QUAL-2E parameters were used as the starting point for SWAT in-stream calibration of only the Kings Creek segment.

The profile or decay plots of total N and total P measured along the 60 km length of Kings Creek was compared with the prediction of SWAT and independent QUAL-2E (figs. 5 and 6). Although the profile of total N did not match well for the middle of the segment, the predicted concentrations were within a reasonable range of measured values. This could be because of excess nitrate contribution from lateral flow simulated by SWAT. As the measured concentrations were within a reasonable range, adjustments were not made to lateral flow without further information and to avoid overfitting the model. The profile of total P seemed to match reasonably well with the measured values (fig. 6). Similar profile plots of ammonia, nitrate + nitrite, organic nitrogen, mineral phosphorus, organic phosphorus, dissolved oxygen, carbonaceous biochemical oxygen demand, and chlorophyll-*a* predicted by SWAT were also compared with the observed data (result is not presented here).

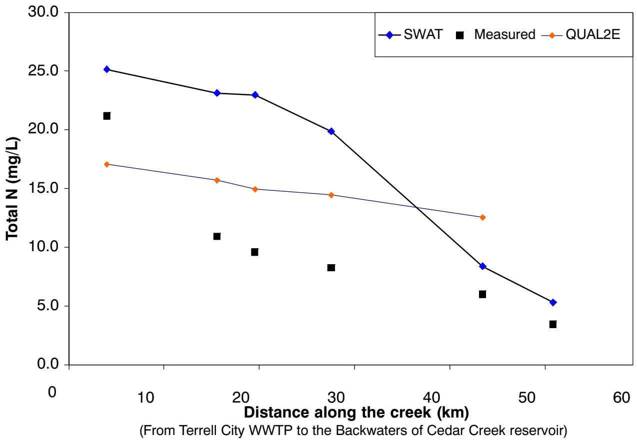

Figure 5. Profile of total N during the 24 h intensive sampling study.

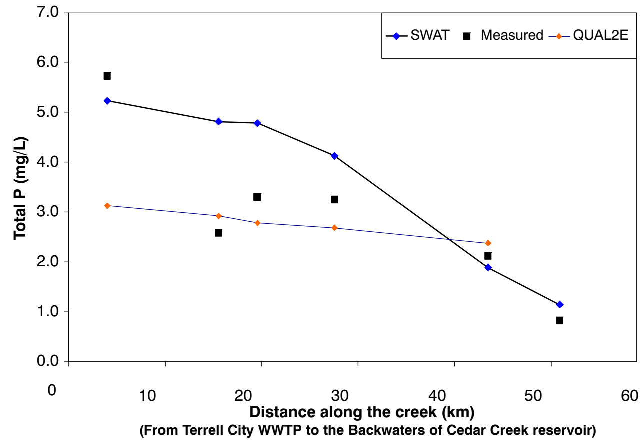

Figure 6. Profile of total P during the 24 h intensive sampling study.

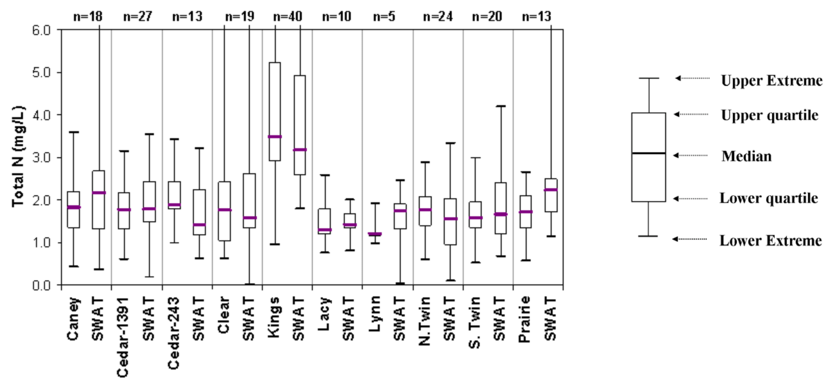

Figure 7. Box plots of observed and predicted (SWAT) values of total N at periodic grab sample locations on major tributaries to Cedar Creek reservoir (sample locations are shown in fig. 1).

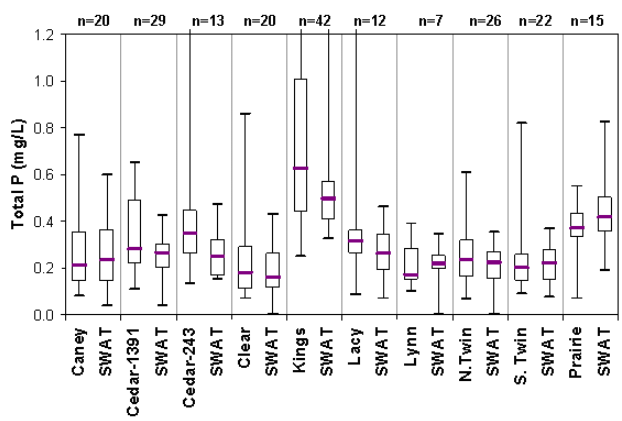

Figure 8. Box plots of observed and predicted (SWAT) values of total P at periodic grab sample locations on major tributaries to Cedar Creek reservoir (sample locations are shown in fig. 1).

In the next step of the calibration, the SWAT default parameters were used as a starting point for the remainder of the subbasins. The simulation period was from 1989 through 2002. The output from this simulation was compared to median water quality data collected by TRWD (grab samples) from 1989 through 2000 in each major tributary. In order to account for the daily variability of SWAT, simulated output was averaged for the three days surrounding the day of the measured grab sample. The coefficients for all subbasins, except those in the Kings Creek watershed that correlate to the 24 h intensive study, were adjusted for each watershed to match the observed data.

Attempts to calibrate on a frequent basis, such as comparing each observed sampling event or even quarterly, were too variable. Rather than matching every individual observation point, our objective was to get a prediction within a reasonable range of observed values. Box plots based on the median, 25th percentile, and 75th percentile of the 3-day averages from SWAT were compared to the box plots of the median, 25th percentile, and 75th percentile of the measured grab samples to adjust the in-stream parameters. Each box plot shows the minimum value (represented by the lower whisker). The box itself represents the middle 50% of the data (bounded by the lower quartile, median, and upper quartile), and the maximum value is shown by the upper whisker. The "n" value on top of each box indicates the number of data points available for calculating the statistics needed for the box plot.

Study of the box plots (figs. 7 and 8) shows that SWAT-simulated total N and total P were within the range of observed values at different sites. Statistical comparison of observed tributary median concentrations from 1989 to 2002 to SWAT predictions for the same period (as shown in figs. 7 and 8) gave statistically significant R^2 values of 0.7 for total N and 0.8 for total P. Similar distribution plots of ammonia, nitrate + nitrite, organic nitrogen, mineral phosphorus, organic phosphorus, dissolved oxygen, carbonaceous biochemical oxygen demand, and chlorophyll-*a* predicted by SWAT were also compared with the observed data for model calibration (result is not presented here). As can be seen from table 2, the QUAL-2E parameters obtained from an independent calibration were only slightly altered within SWAT to do the nutrient calibration.

Finer Aspects of Model Calibration

Having an accurate flow simulation is an essential precursor to making a realistic prediction of sediment and nutrient loading from the watershed. Hence, the flow was first calibrated followed by the sediment and then the nutrients. However, while doing the sediment calibration, adjusting watershed parameters such as initial soil residue cover (RSDIN) and minimum USLE cover and management factor (USLE_C) also has an indirect impact on the runoff simulation. This is because organic nutrients are attached to the soil particles and transported with surface runoff. This loss of nutrients due to erosion affects the soil nutrient balance and hence the plant growth, which in turn affects evapotranspiration, antecedent soil moisture conditions, and runoff. Similarly, adjusting watershed parameters such as mineralization of organic nitrogen (CMN), nitrogen and phosphorus uptake factors (UBN and UBP), phosphorus availability index (PSP), etc., will also affect the runoff simulation.

Hence, flow, sediment, and nutrient calibration was done in an iterative process, where the flow was first calibrated to a reasonable level by adjusting only the coefficients related to flow (table 1) followed by iterative adjustments of flow and sediment parameters. The iterative adjustments were done until the overland erosion values predicted by the model for different landuses were reasonable based on field survey and/or reported literature values. Following a reasonable calibration of flow and overland erosion, nutrient calibration was done in a similar iterative loop until the predicted in-stream nutrient concentrations were within the range of observed values. Further, fine-tuning of in-stream water quality parameters (table 2) was needed to calibrate the water quality simulations.

Reservoir Water Quality Calibration

The reservoir water quality model WASP was set up based on the nonpoint and point source loading simulated from the watershed. Refer to Ernst and Owens (2009) for a detailed description of WASP model setup for Cedar Creek and its calibration. The WASP model was calibrated for an 11-year period from 1991 to 2001. Comparison of median (both annual and seasonal) observed and predicted data for the 11-year WASP model at seven segments in the reservoir showed an R^2 value greater than 0.9 for total N and total P, indicating a good calibration of the lake model.

Spatial Distribution of Sediment Loads

The distribution of sediment loading by each land cover category is given in figure 9. It clearly shows that even though cropland occupies only 6% of the total land cover, it contributed more than 41% of the total sediment loading in the basin. The average annual erosion rate of cropland is about 12 metric tons per hectare (t ha^{-1}) and varies from less than 1 t ha^{-1} to over 63 t ha^{-1} (equivalent to losing about 0.1 to 4.3 mm of top soil every year, with a bulk density of 1.45 g cm^{-3}). Croplands are mostly concentrated in the northern portion of the watershed. Significant urban development is also occurring in this region. Next to cropland, urban landuse has the next highest erosion rate (2 t ha^{-1} per year). Hence, subbasins in this portion of the watershed have high erosion rates when compared to other subbasins (fig. 10). Hence, sediment control BMPs should be considered in heavily eroding croplands and urban lands. As the pasturelands were only in fair hydrologic condition, they contributed over 15% of the total sediment (0.45 t ha^{-1}). However, if they were maintained in good crop cover condition, the erosion from the pasturelands would be much smaller.

For the soils in Cedar Creek watershed, the soil loss tolerance value (T-value from SSURGO database) is between 6.7 and 11.2 t ha^{-1} (3 to 5 tons per acre). The T-value indicates the maximum tolerable soil loss that could take place without causing significant decline in long-term productivity. As can be seen from the SWAT-simulated values, the cropland is eroding at a rate slightly higher than this tolerance value and hence needs to adopt conservation measures to reduce the rate of overland erosion and improve soil productivity.

The second major contributor of sediment is channel erosion (34%) (fig. 9). The rapid geomorphic survey of the channels across the watershed provided the much-needed input to adjust the SWAT parameters related to channel erosion. The average annual channel erosion for each channel segment was normalized based on its length and cross-sectional area for uniform comparison across the watershed. For a unit length (1 km) of channel, most of the stream segments are eroding at a rate of 12 to 36 metric tons per square meter of channel cross-section (fig. 10). BMPs such as riparian buffers and channel stabilization structures could be considered on heavily eroding stream segments.

Sediment mass balance based on the total sediment generated in the watershed and the actual amount reaching the lake showed that about 97% of the sediment generated in the watershed is reaching the reservoir. Only 3% of the sediments are settling in the channels. The field survey also confirmed that most of the stream channels are degrading, and only a very few stream segments close to the lake are aggrading due to sediment deposition.

Spatial Distribution of Nutrient Loads

Analysis of the nutrient loading into the lake showed that the majority (about 65%) of the phosphorus and nitrogen are in organic form. Sediment plays a major role in the transport of these nutrients. Hence, the distribution of nutrients by landuse and their spatial distribution are similar to that of sediments (figs. 9 and 10). The WWTP contributes about 7% and 12% of total N and total P loads, respectively. Significantly, channel erosion is also estimated to produce nutrients loads equivalent to that of WWTP. This estimation was possible due to a simple modification made to the in-stream mass balance of nutrients based on channel erosion, as described previously.

Nutrient mass balance showed that only about 86% of the total N and 87% of the total P generated in the watershed reaches the lake. The reminder settles to the channel bottom. This is slightly anomalous because more than 97% of the sediment reaches the lake, and nutrient settling is primarily due to settling of sediment materials. The reason for this anomaly is because SWAT routes sediment and nutrients in separate pools and not as bounded units. During model calibration, the

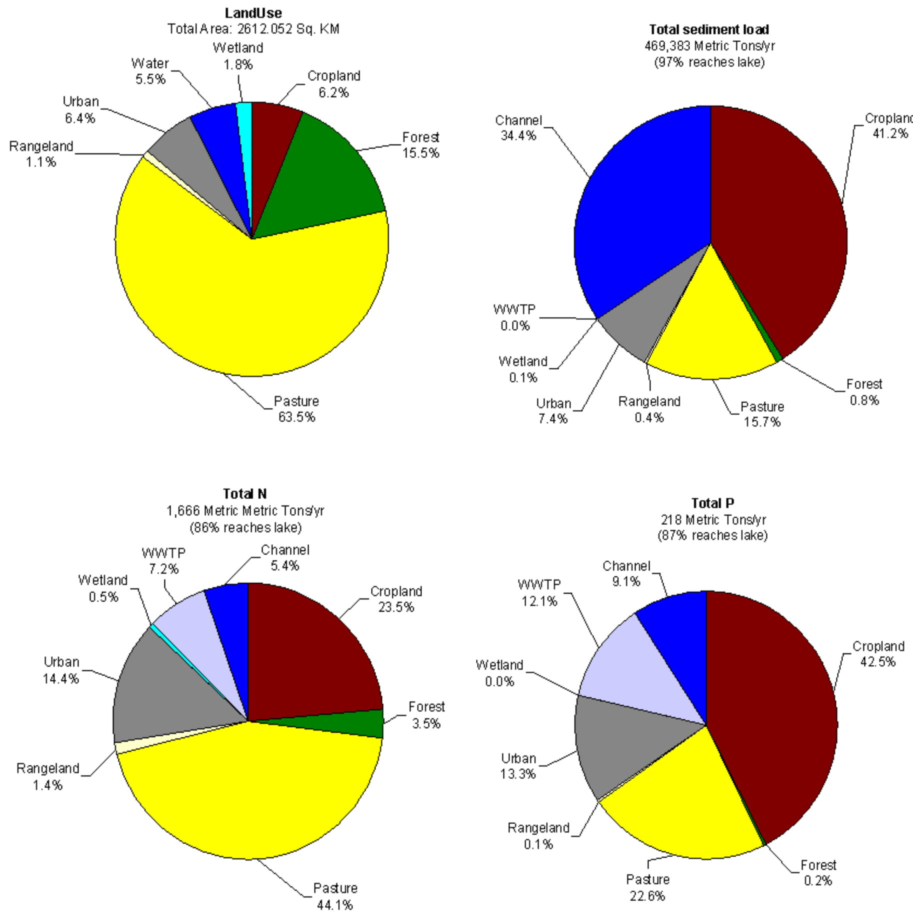

Figure 9. Sediment and nutrient load distribution among various land use categories based on SWAT model simulation.

settling rate parameters of nutrients were adjusted based on the observed water quality data. This discrepancy cannot be avoided completely unless both sediments and nutrients are routed together as bounded units. This could be a future improvement to the in-stream modeling routines in SWAT.

Sediment and Nutrient Hot Spots

Based on the model results (fig. 10), it is clear that the northern portion (Kings Creek) of the watershed has several subbasins that are eroding at a much higher rate than the basin average rate of 1.18 t ha^{-1} per year. Nine subbasins are eroding at a rate between 2.63 and 7 t ha^{-1} per year (fig. 10). Hence, these subbasins also contribute a considerably higher amount of the total N and total P load when compared to the rest of the subbasins. The higher erosion rate of these subbasins is because of the higher percentage of cropland and urban area in these locations. Hence, as noted before, special emphasis must be given to agricultural and urban best management practices, such as contour farming, crop residue management, pasture planting, and fertilizer and nutrient management that reduce soil erosion and nutrient loads into Cedar Creek reservoir. However, cropland or urban BMPs alone will not be sufficient to improve the lake water quality. A combination of BMPs that improve the pastureland, rangeland, and channels should also be developed and adopted in an integrated way to improve the lake water quality.

Load Reduction Scenarios to Improve the Lake Water Quality

A mass balance analysis of nutrients entering the lake showed that about 86% of the total N load into the lake comes from the watershed (nonpoint source + seven WWTPs in the watershed). About 5% of the load comes from the two lakeside WWTPs, 7% from the atmosphere, and 2% from the lake bottom benthic flux. For total P, about 87% of the load comes from the watershed, 7% comes from the lake side WWTP, 3% from the atmosphere, and 3% from the lake bottom. The sensitivity of the chl'*a*' concentration predicted by the calibrated WASP model to each of the four sources of nutrients was evaluated independently by systematically shutting off each load. The response of algae (chl'*a*') growth during the calibration period for segment 4 is presented in figure 11, where statistical testing with a Kruskal-Wallis multiple comparison test ($\alpha = 0.05$) shows all simulations that are not significantly different from the calibration as having the same letter designation (i.e., A). These results suggest that the watershed loading is the most important contributor of nutrients necessary for the algal growth in the lake. As the watershed loading is the significant contributor to chl'*a*' growth, five nonpoint source load reduction scenarios were simulated. A statistically significant reduction in chl'*a*' was achieved only when the nonpoint source load (both total N and total P) from the watershed was reduced by at least 35% or more (fig. 12).

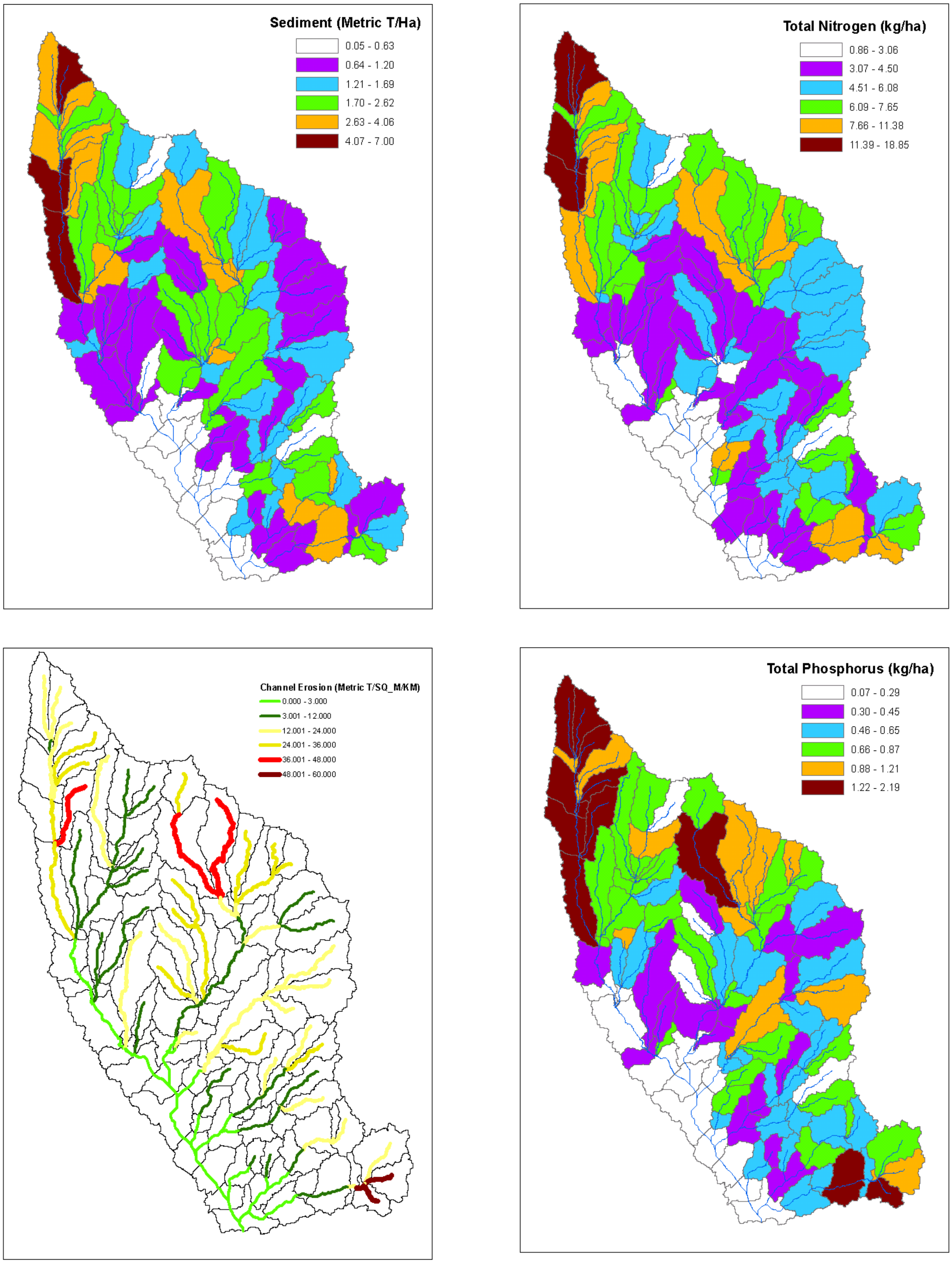

Figure 10. Spatial distribution of sediment and nutrient loads.

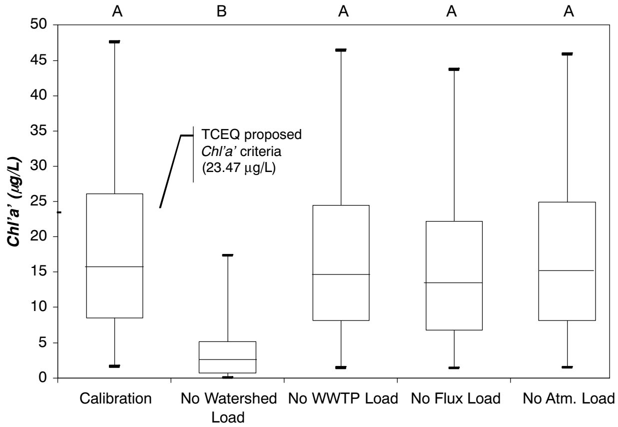

Figure 11. Sensitivity of chl'*a*' to systematic removal of each nutrient load source (box plots represent median and 25th and 75th percentiles. Box whiskers labeled with the same letter are not significantly different at p = 0.05.)

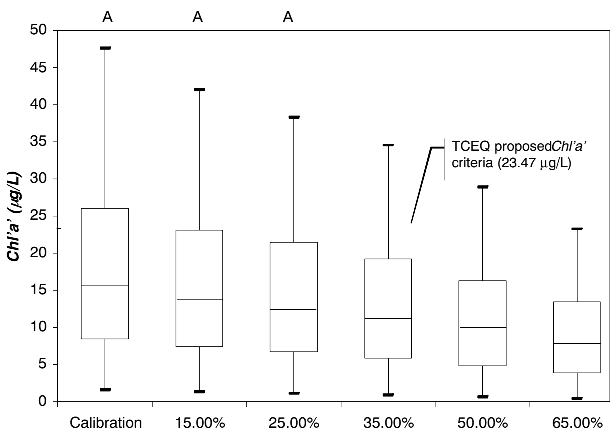

Figure 12. Reduction in chl'*a*' for different levels of nonpoint source load reduction (box plots represent median and 25th and 75th percentiles. Box whiskers labeled with the same letter are not significantly different at p = 0.05.)

Preliminary SWAT model runs of individual BMPs for nonpoint source loads from cropland, pasture, urban land, and channels show that a sediment load reduction of 5% to 28%, a total N reduction of 1% to 18%, and a total P reduction of 1% to 35% are feasible. Conversion of all cropland to pasture and urban nutrient management (reducing lawn fertilization) gave the most significant load reduction (Narasimhan et al., 2008). However, any single BMP alone may not be feasible for watershed-wide implementation nor technically sufficient to achieve a nonpoint source load reduction of about 35%. Hence, an economic study is currently underway to identify a suite of cost-effective BMPs that would reduce the nonpoint source load into the lake by 35%.

Conclusion

A comprehensive modeling approach has been developed in which the watershed model SWAT was linked with the lake eutrophication model WASP for developing management options to maintain the lake water quality of Cedar Creek reservoir in north central Texas. SWAT provides critical spatial and temporal hydrology and loading information to drive WASP, which in turn provides detailed spatial and temporal eutrophication insights for the lake. Linking the watershed model with the lake water quality model enables us to understand the dynamics of nutrient loading from the watershed in large reservoirs such as Cedar Creek reservoir. Channel erosion within a large basin such as Cedar Creek could be a significant component. Hence, nutrient loading from channels should also be considered while developing the watershed management plans.

Based on the spatially distributed watershed model, subbasins that contribute sediments and nutrients have been identified. Several best management practices are currently being assessed with SWAT, and an economic study is underway to come up with a suite of cost-effective watershed management practices that will reduce the watershed load by about 35% (total N and total P), primarily from the identified subbasins in the watershed.

Acknowledgements

The authors wish to acknowledge the support of Woody Frossard, Darrel Andrews, and Jennifer Owens of Tarrant Regional Water District, Fort Worth, Texas; David Harkin of Espey Consultants; and Alan Plummer of Alan Plummer Associates for their valuable suggestions in modeling. Many thanks to Clint Wolfe, David Waidler, and Dr. Allan Jones of Texas AgriLife Research and Extension Urban Solutions Center, Dallas, Texas, for their unflinching support in coordinating the project. We thank the three anonymous reviewers for their constructive comments, which improved the clarity of this publication.

References

Allen, P. M., B. Narasimhan, J. Dunbar, S. Capello, and R. Srinivasan. 2007. Rapid geomorphic assessment of watershed sediment budgets for water supply reservoirs using SWAT and sub-bottom acoustical profiling. In *Proc. 4th Intl. SWAT Conf.*, 63-72. Delft, The Netherlands: UNESCO-IHE.

Arnold, J. G., P. M. Allen, R. Muttiah, and G. Bernhardt. 1995. Automated base flow separation and recession analysis techniques. *Ground Water* 33(6): 1010-1018.

Bagnold, R. A. 1977. Bedload transport in natural rivers. *Water Resources Res.* 13(2): 303-312.

Cerucci, M., and G. K. Jaligama. 2008. Hydrologic and water quality integration tool: HydroWAMIT. *J. Environ. Eng.* 134(8): 600-609.

Debele, B., R. Srinivasan, and J. Yves Parlange. 2006. Coupling upland watershed and downstream waterbody hydrodynamic and water quality models (SWAT and CE-QUAL-W2) for better water resources management in complex river basins. *Environ. Model. and Assess.* 13(1): 135-153.

Dodds, W. K., W. W. Bouska, J. L. Eitzmann, T. J. Pilger, K. L. Pitts, A. J. Riley, J. T. Schloesser, and D. J. Thornbrugh. 2009. Eutrophication of U.S. freshwaters: Analysis of potential economic damages. *Environ. Sci. and Tech.* 43(1): 12-19.

Ernst, M. R., and J. Owens. 2009. Development and application of a WASP model on a large Texas reservoir to assess eutrophication control. *Lake and Reservoir Mgmt.* 25(2): 136-148.

Narasimhan, B., P. M. Allen, R. Srinivasan, S. T. Bednarz, J. G. Arnold, and J. A. Dunbar. 2007. Streambank erosion and best management practice simulation using SWAT. In *Proc. 4th Conf. on Watershed Management to Meet Water Quality Standards and Emerging TMDLs.* ASABE Paper No. 701P0207. St. Joseph, Mich.: ASABE.

Narasimhan, B., T. Lee, and R. Srinivasan. 2008. Cedar Creek watershed: Best management practice development and simulation using SWAT. Fort Worth, Tex.: Tarrant Regional Water District.

Nash, J. E., and J. V. Sutcliffe. 1970. River flow forecasting through conceptual models: Part 1. A discussion of principles. *J. Hydrol.* 10(3): 282-290.

Santhi, C., J. G. Arnold, J. R. Williams, W. A. Dugas, R. Srinivasan, and L. M. Hauck. 2001. Validation of the SWAT model on a large river basin with point and nonpoint sources. *J. American Water Resources Assoc.* 37(5): 1169-1188.

USEPA. 1996. Environmental indicators of water quality in the United States. Washington, D.C.: U.S. Environmental Protection Agency.

Williams, J. R. 1980. SPNM, a model for predicting sediment, phosphorus, and nitrogen yields from agricultural basins. *Water Resour. Bull.* 16(5): 843-848.

Windhorn, R. H. 2001. RAP-M: Rapid Assessment Point Method. Champaign, Ill.: USDA-NRCS.

Wool, T. A., J. L. Martin, and R. W. Schottman. 1994. The linked watershed/waterbody model (LWWM): A watershed management modeling system. *Lake and Reservoir Mgmt.* 9(2): 124.

Wu, J., S. L. Yu, and R. Zou. 2006. A water quality-based approach for watershed wide BMP strategies. *J. American Water Resources Assoc.* 42(5): 1193-1204.

Xu, Z., A. N. Godrej, and T. J. Grizzard. 2007. The hydrological calibration and validation of a complexly linked watershed-reservoir model for the Occoquan watershed, Virginia. *J. Hydrol.* 345(3-4): 167-183.